
**Market-based
Approaches to
Environmental
Regulation**

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Market-based Approaches to Environmental Regulation

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Abstract

Economists argue that policymakers should take advantage of market principles in designing environmental regulations. Such market-based approaches use economic incentives to achieve environmental goals at lower costs including taxes and “cap-and-trade”. While these economic approaches to environmental problems have existed for many years, they have been slow to be adopted as actual regulations. Market-based approaches have now become more common due to advocacy by economists and early positive policy experiences. Despite this acceptance, a gap remains between real-world market-based policies and the ideas propounded by academic economists. Policymakers have often merged market-based incentives onto existing non-market approaches resulting in a set of mixed policies whose economic properties are often difficult to unravel. The economics of such mixed approaches remains under-explored. A similar sort of gap exists between the problems that market-based regulations should tackle and the problems that they actually address. The authors review the economics literature of market-based environmental regulations and design issues for environmental taxes and cap-and-trade systems.

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1

Introduction

For nearly a century, economists have argued that policymakers should take advantage of market principles in designing environmental regulations. Such market-based approaches would use economic incentives to achieve environmental goals at lower costs. Pigou (1920) suggested levying a tax on production activities that generate environmental externalities and showed that this would achieve the same desirable effects as the free market does for ordinary goods. Much later, Dales (1968) suggested that the same advantages could be gained if polluters were assigned transferable rights to their pollution, with the total number of such rights set equal to the overall emissions goal. This approach to environmental regulation was originally known as tradable permits and is now known simply as cap-and-trade. These two mechanisms – taxes and cap-and-trade – together constitute the set of market mechanisms.

While these economic approaches to environmental problems have existed for many years in the minds of economists, they have been slow to be adopted as actual regulations. The Clean Air Act Amendments of 1970 and the Clean Water Act of 1972 – the cornerstones of U.S. pollution policy – are devoid of regulations based on economic incentives as recommended by the economics literature. Shortly after their passage, however, policymakers began to experiment with market-oriented solutions.

2 *Introduction*

Market-based approaches have now become more common, due in large part to the long-standing and unanimous advocacy of such approaches by economists and some early positive policy experiences. Market-based approaches have become more widely accepted among policymakers as reasonable ways to tackle U.S. environmental concerns. Even among environmentalists, support for market-based approaches has increased, although many critics still exist in this community.

Despite this apparent acceptance, a gap remains between the real-world market-based policies that have made it into law and the ideas that have been propounded by academic economists over the course of eighty-plus years. In short, policymakers have often grafted or merged market-based incentives onto existing non-market approaches. The result is a set of mixed policies whose economic properties are often difficult to unravel. Economists have almost uniformly conceived of market approaches as the sole regulatory instruments to be used for a given problem, but this has rarely been the case in practice. Careful examination of any environmental regulation, even ones considered by most people as market-based, will reveal that they in fact consist of a complicated mix of market and non-market mechanisms. The economics of such mixed approaches remains under-explored.

A similar sort of gap exists between the problems that market-based regulations should tackle and the problems that they actually tackle. Since the most common market instrument is cap-and-trade, we call this the “misplaced cap” problem. Fuel standards are a perfect example; they cap miles per gallon of new cars, not gallons of gasoline or miles driven, which are closer to true externality causes. As with the mixed approach, the economics of this misplaced cap problem have received rather little attention.

In Section 2, we review the economics literature on the theory of market-based environmental regulations. Section 3 covers design issues for environmental taxes and cap-and-trade systems. In Section 4, we discuss the U.S. experience with a number of regulatory approaches that are commonly characterized as market-based. We describe the mix of market and non-market instruments that characterize these policies. Section 5 draws our main conclusions.

2

Environmental Regulation and the Case for Market Mechanisms

2.1 The neoclassical argument for taxes and cap-and-trade

Before we discuss the economics of regulation, we reiterate why any such government intervention is necessary in the first place. In a market system, prices are used to coordinate the activities of buyers and sellers and to convey information about the strength of consumer demand for a good and the costs of supplying it. Because trade is voluntary, buyers and sellers only make exchanges when both parties benefit, so government intervention in such an economy is not needed to increase these parties' welfare.

A problem – known as an *externality* – arises, however, when a voluntary transaction between parties affects a third party, whether positively or negatively. For example, a firm might produce and sell a good to a consumer to both the firm's and the consumer's advantage, but the production may result in air pollution that reduces the health of other people living near the firm. When externalities exist, the market system will typically lead to an inefficient outcome because the impact on any third parties is not considered by the parties participating in the trade. The market failure of pollution and other environmental

externalities therefore stems from the freedom of firms, consumers, and other agents to “use” the environment without restraint, in the absence of regulation.

The root of this market failure is that there is no clear property right for the environmental good (Coase, 1960). If society could create a property right for, say, “clean air,” to be held by individual citizens, then any action that involved polluting the air would require the consent of the right’s owner and therefore would necessarily compensate her for any disutility. Similarly, if the polluter owned the right to pollute the air, then individuals would need to pay the firm to reduce its pollution. That is, depending on who owned the property right, either polluters would need to purchase the use of the air or the victims of the pollution would need to pay polluters to reduce pollution.

However, because of its public good nature, the normal channels by which society creates and enforces property rights are largely absent for the environment. One person cannot sell her clean air rights without simultaneously affecting the air quality experienced by everyone else in the vicinity. This means that any transaction regarding clean air rights would affect the air quality experienced by third-parties not participating in the transaction. Those third party’s interests should be brought into consideration. Their inclusion can occur either through multi-party bargaining or through the government stepping in to represent their interests. Multi-party bargaining is particularly difficult when the externality affects a large number of people, leaving government action as the principal remedy. These multi-party issues do not generally arise for private goods because their production and trading only affect the parties directly involved.

The work of Coase thus provides a key to the protection of environmental quality. The government should try to make the environment “look like” a private good to its users. Since the lack of a price for the environment leads to its inefficient overuse or over-consumption, the heart of efficient regulation is the creation of a “price” for pollution.¹

¹Throughout this article, we use pollution or emissions to stand for any activity that reduces environmental quality, including deforestation or over-fishing, for example. Likewise, we use firm or polluter to stand for any agent who undertakes an environment-affecting activity. Our arguments are general and should not be construed to apply only to pollution or

This price-creating property is the essential characteristic of a market instrument. Furthermore, the ideal price should represent the interests of all parties who are affected by environmental quality.

The generation of a market price for pollution is achieved by one of two means: Either the price is set directly through a pollution tax or indirectly through establishment of a cap-and-trade system. As Pigou (1920) first argued, a tax on pollution provides an incentive for a polluting source to reduce its pollution and thereby economize on its use of the environment.² For each unit of pollution, the polluter must choose either to pay the tax or reduce that pollution through any means at its disposal. Each source will reduce its emissions, relative to the “free” pollution case, until it costs more (in terms of lost profits or higher abatement costs) to reduce a unit of emissions than to pay the emissions tax. This results in marginal abatement costs that are equal to the tax and therefore equal across all regulated sources. The tax should be set at a level that achieves the desired level of emissions. Environmental taxes are also known as environmental fees or charges or, more euphoniously, green taxes.

The second approach, suggested by Dales (1968), requires each regulated source to submit a permit (also known as a quota, credit, or allowance) for each unit of pollution emitted. These permits are transferable, thus allowing different pollution levels across the regulated entities. By allocating fewer permits than the “free” pollution level, the regulatory agency creates a shortage of permits which then leads to a positive price for permits. This establishes a price for pollution, just as in the tax case.

Similarly to the pollution tax case, each source will reduce its emissions until it costs more for the source to reduce one more unit of emissions than to buy (or not sell) a permit. If the permit market is perfectly competitive, then marginal abatement costs will be equal to the permit price and therefore equal across all regulated sources. The equality of marginal abatement costs is a necessary condition for

only to environmental damages caused by production. For a discussion of the definition of a (negative) externality, which is the essence of our discussion, see Baumol (1972).

²Pigou wrote in terms of a tax on output, but his assumptions made this identical to a tax on pollution.

any given level of environmental quality to be achieved at the lowest overall cost, a condition known as cost-effectiveness. This is, in turn, a necessary condition for Pareto efficiency. The number of permits that the regulatory agency issues should be the desired level of emissions.

To see that taxes are cost-effective, let r_k represent a vector of inputs used by producer $k = 1, \dots, K$; w , the price vector of inputs; s_k , the pollution generated by plant k ; and $f^k(r_k, s_k)$ the plant's output. Let p represent the price of this output. Then the cost-effective choice of inputs $r = \{r_1, \dots, r_K\}$ to generate overall pollution no greater than S satisfies:

$$\max_r \sum_k p f^k(r_k, s_k) - w r_k \quad \text{subject to} \quad \sum_k s_k \leq S \quad (2.1)$$

For a well-behaved problem, this yields first order conditions:

$$p \frac{\partial f^k}{\partial r_{ik}} - w = 0, \quad p \frac{\partial f^k}{\partial s_k} - \lambda = 0 \quad (2.2)$$

where λ is the Lagrange multiplier pertaining to the aggregate pollution equation.

Next consider the case of a pollution tax. The model's key element is the production decisions of the individual polluters who face pollution tax τ . Each firm's objective function is to maximize $p f^k(r_k, s_k) - w r_k - \tau s_k$. This yields first order conditions:

$$p \frac{\partial f^k}{\partial r_{ik}} - w = 0, \quad p \frac{\partial f^k}{\partial s_k} - \tau = 0 \quad (2.3)$$

and these are equivalent whenever $\tau = \lambda$. Alternatively, for any given τ , there exists an S^* such that $\lambda(S^*) = \tau$. Thus, a pollution tax is always cost-effective; whatever pollution amount it yields will have been generated at the lowest cost (measured in foregone profits) to the economy. Pollution tax revenues do not appear in the model because they are assumed to be recycled in the economy in a neutral way.

For the case of tradable permits, let S^* be the aggregate number of permits issued and let S^k be k 's permit endowment, with $\sum_k S^k = S^*$. This endowment is assumed to be exogenous to the firm. The price of permits, ρ , is endogenous to the market but assumed exogenous to any individual polluter. For a given ρ , the firm's objective function

is to maximize $pf^k(r_k, s_k) - wr_k - \rho(s_k - S^k)$. The first order conditions are:

$$p \frac{\partial f^k}{\partial r_{ik}} - w = 0, \quad p \frac{\partial f^k}{\partial s_k} - \rho = 0 \quad (2.4)$$

Note that K versions of equation (2.4) hold, where K is the number of polluters. These K equations plus $\sum_k S^k = S^*$ yield $K + 1$ equations in $K + 1$ unknowns, s_1, \dots, s_K and ρ , thus yielding the market-clearing permit price. From (2.2) and (2.4) it is then clear that $\rho = \lambda$.

The equivalence of equations (2.2), (2.3), and (2.4) is the essence of the neoclassical argument for taxes and cap-and-trade. Of course, the theoretical results are conditional on numerous assumptions, many of which are covered in a qualitative sense in Section 3. The failure of any such assumption does not, however, obscure the basic message of (2.2), (2.3), and (2.4). For further demonstration and discussion of these and related claims, see Baumol (1972); Cornes and Sandler (1986); Baumol and Oates (1988); and Cropper and Oates (1992). Baumol and Oates (1988) in particular provides a thorough discussion of circumstances where equations (2.2) or (2.3) are solutions to local rather than global maxima.

This model does not explicitly incorporate the fact that regulators will almost always be less knowledgeable about pollution control costs and the desirability of various pollution control strategies than the individual firms will be, the so-called agency problem. The recognition of this problem underlies this literature, however. Although a regulator could conceivably directly specify the pollution levels of each firm or even the individual input and output bundles, the early developers of environmental policy models recognized that this approach was unrealistic and unadvisable, even if the models did not explicitly capture this sentiment. The agency problem was incorporated implicitly through the accompanying discussion and interpretation of the models and results.

Models explicitly incorporating asymmetric information and a principal-agent framework for environmental policy began in the late 1970s and early 1980s; Dasgupta et al. (1979) appears to be the first of these. These models focused on the more difficult problem of achieving

the socially optimal level of pollution, which includes the external damages from pollution.

The next step beyond cost-effectiveness is that the level of pollution itself, S , should be Pareto-efficient. An efficient tax or cap-and-trade system should yield a price for pollution equal to its marginal external cost. To demonstrate this in the context of the above model, we adopt a more modern approach in which pollution damages are monetized and then expressed as $D(S)$. $D(S)$ represents, roughly speaking, the external costs imposed on the economy as a whole when the pollution level is S . The efficient level of pollution thus solves:

$$\max_{r,s} \sum_k p f^k(r_k, s_k) - w r_k - D\left(\sum s_k\right) \quad (2.5)$$

This set-up assumes a constant price for the produced goods, so there is no consumer surplus (or change in consumer surplus). For a more general and more rigorous treatment, see Baumol and Oates (1988). The first order conditions are now given by:

$$p \frac{\partial f^k}{\partial r_{ik}} - w = 0, \quad p \frac{\partial f^k}{\partial s_k} - D'\left(\sum s_k\right) = 0 \quad (2.6)$$

A comparison of (2.6) with (2.3) or (2.4) now shows the optimal tax or permit level. Let S^* solve (2.6). In the case of taxes, the optimal tax is equal to the marginal pollution damage, $\tau = D'(S^*)$.³ In the case of cap-and-trade, the optimal permit level is S^* and the permit price will be $\rho = D'(S^*)$, which is equal to the optimal tax rate. (For a more thorough treatment of the set of necessary conditions for Pareto efficiency without the monetization of pollution damages, see Baumol and Oates, 1988).

Setting the tax or permit cap to satisfy (2.6) requires knowledge of the marginal damages caused by emissions. However, even when this knowledge is absent, taxes and cap-and-trade are still able to achieve the resulting environmental outcome at the lowest overall cost. Indeed,

³Technically, this result depends on perfect efficiency in how tax revenue is disposed of, including tax revenue from other sources (which could be offset by pollution tax revenue). Under the perfect-efficiency assumption, the marginal benefit or cost of additional government tax revenue is zero, so tax revenue disappears from the model, leaving just profits minus damages. See Section 3.5.

the argument for market mechanisms is that they are *uniquely* able to achieve this “lowest overall cost” criterion.⁴

The agency problem is more acute when S is required to be Pareto-efficient. The problems can be demonstrated even in the context of models as restrictive as (2.5), since the optimal tax or cap-and-trade now generally requires knowledge of s_k^* , which in turn requires knowledge of individual firm costs. Only when $D(S)$ is linear, so that $D'(S)$ is constant, does the agency problem disappear.

There is one additional point discussed extensively in Baumol and Oates (1988). Pollution damages depend in part on individuals’ defensive measures taken against the pollution exposure. The above equation for optimality is contingent on each consumer choosing an optimal level of protection. Note that if the regulatory authority uses the revenue gained from the pollution tax to compensate the victims of the pollution, then this would distort the optimal expenditure on defensive actions, resulting in inefficiency. In other words, compensating the victims of externalities will result in a moral hazard problem where the victims do not engage in appropriate levels of defensive actions.

2.2 The cost-savings argument for taxes and cap-and-trade

This section presents the non-technical, policymakers’ argument for why the two market mechanisms are considered superior to alternative regulatory approaches to environmental problems. Their superiority stems from the *flexibility* they bestow on sources over how to control pollution or other environmentally-damaging actions. Market mechanisms provide both within-firm and across-firm flexibility, flexibilities that are missing from non-market regulations. Such flexibility ensures that the lowest-cost pollution control actions are taken. We reiterate our admonition that pollution is used by us to refer to environmentally damaging activities of any sort.

⁴If the regulator had perfect information about abatement costs *and* perfect ability to specify abatement actions, then a third approach – direct specification of pollution control actions – would also achieve lowest-cost control. We consider these conditions to be essentially impossible.

Consider first the role of within-firm flexibility. Since the costs of controlling pollution will be borne entirely by the polluter, the polluter has every incentive to take any action whose cost is less than the tax or permit price. The greater the within-firm flexibility the source has, the wider the range of options it can consider. The wider the range of options considered, the lower will be the costs of pollution control. “Cost” is used here in a very general sense; it refers not only to out-of-pocket costs but also to time, disutility, and agency costs. So long as these costs are borne entirely by the polluter then within-firm flexibility will lead to the lowest-cost actions being taken by each polluter.

Consider next the role of across-firm flexibility. Across-firm flexibility allows different levels of pollution among the different regulated sources. With across-firm flexibility, a source may choose not only from among its own possible actions to reduce emissions but also, implicitly, from *other sources’* possible actions. Costs are lower than under within-firm flexibility alone because sources will search for the lowest cost actions across the entire regulated sector.

Market instruments exhibit a powerful form of across-firm flexibility. With a single “price” for environmental damage, as provided by taxes or cap-and-trade, the marginal cost of the last unit of pollution reduction will be equal across sources, thus minimizing the total cost of the regulation (see Dales, 1968; Montgomery, 1972; Baumol and Oates, 1988). In less technical terms, market regulations provide *complete* across-firm flexibility. Any pollution control opportunity anywhere (that is covered by the regulation) will be exploited by taxes or cap-and-trade.

An essential element of this demonstration of cost-effectiveness is that the tax or permit requirement is assumed to be the *sole* legal restriction on emissions. Under a true market mechanism, other restrictions – whether on pollution control technology, emissions rates, energy efficiency, or any of a host of restrictions that have been adopted or proposed over the years – are both unnecessary and unwarranted. They are unnecessary because under a market mechanism the incentive not to incur the tax or not to have to purchase more permits will already have led polluters to find and adopt all appropriate actions. They are unwarranted because those restrictions may not be the lowest cost strategy

for the source to reduce its emissions. If those restrictions were part of the lowest cost strategy, then there would be no need to impose them; market forces would lead polluters to find and adopt them. If they were not part of the lowest cost strategy, then they should not be imposed.

Flexibility, of both varieties, may be especially valuable over time. Flexibility allows sources to search not only among currently available control options but among to-be-discovered control options; that is, to undertake research and development to find new technologies.

The policymakers' case for taxes or cap-and-trade is completed by examining the weaknesses of alternative approaches, which are, by definition, non-market. Non-market mechanisms, often lumped together as *command-and-control*, have been the traditional approach for addressing environmental problems. While the general idea behind command-and-control is that the regulated entities are given little discretion in their pollution control efforts, the rubric of command-and-control covers a wide range of regulations with varying degrees of flexibility and cost savings.

Non-market regulations can be roughly categorized as technology or performance regulations.⁵ A *technology standard* is one that prescribes particular actions with little or no flexibility to choose other actions that might lead to the same environmental result. In the case of air pollution, technology standards require the installation of certain control technologies such as scrubbers, with no flexibility for the source to reduce emissions through other means such as switching to a cleaner-burning fuel, reducing production, or devising other innovative pollution-reduction strategies.

In the case of non-point water pollution, a different sort of technology standard applies. To improve water quality, various state and federal programs pay landowners to create stream buffers or to fence streams to keep livestock out. These payments are accompanied by contracts with the landowners; those contracts specify, in the latter case, the number of strands of barbwire, the distance between strands, and the distance between fence posts, among other things. Again, these

⁵Technology-based standards are a form of performance standard, although in practice they often take the form of a technology standard. Note that the economic terms for these regulations do not always correspond with the legal ones or the vernacular.

specifications, even if warranted, do not allow flexibility in meeting a particular water quality goal. This focus on process or technology rather than the environmental outcome is what characterizes a technology standard.

There may be reasons, of course, why a technology standard rather than an alternative approach has been adopted in a particular situation. It seems clear in the non-point pollution case that fence-building rules can be relatively easily monitored but stream pollution or erosion cannot, so such rules may indeed be desirable. However, when the environmental outcome can be measured, either in terms of pollution loadings or stream quality, a regulation that addressed loadings or stream quality directly would almost surely be superior because of the flexibility it would implicitly allow. Even when the environmental outcome cannot be perfectly measured, a regulation that addresses the actual environmental quality may often be superior to a technology standard.

In contrast, *performance standards* focus on pollution or environmental quality and therefore allow regulated sources some degree of flexibility in meeting those goals. Examples of performance standards include plant-level caps (tons of biochemical oxygen demand (BOD), a conventional water pollutant, released per month by sewage treatment plants), emissions per unit of heat input (pounds of sulfur-dioxide per million BTUs) or per unit of output (pounds of mercury per mWh), effluent concentrations (average BOD concentration in effluent), and energy-efficiency standards. These standards exhibit within-firm flexibility because the source is free to choose the best method for achieving the rate or cap. In each case, the regulation focuses on an outcome rather than a set of actions.

Performance standards are cheaper than technology standards but more expensive than market instruments because they lack across-firm flexibility. Energy-efficiency standards on new air conditioners, for example, do not allow some users to buy inefficient air conditioners (presumably because those models have other desirable features), and other users to buy ones that are super-efficient. (The pollution in this case can be thought of either as air pollution or carbon emissions from electricity generation.)

Performance standards in the form of rates (pollution per unit of output or energy efficiency are two common examples) are far more common than performance standards in the form of plant-level pollution caps but are also less efficient because they do not provide an incentive to reduce emissions through reductions in output.⁶ In the case of air conditioners, energy-efficiency standards do not give purchasers any incentive to reduce the number or size of air conditioners that might be purchased or how much each of them will be run; only the energy-efficiency of a particular-sized model is regulated. Similarly, a limit on nitrogen oxides emissions per mile for new automobiles does not give any incentive to reduce overall air pollution by reducing the number of cars or the number of miles driven.⁷

Performance standards that take the form of plant-level caps are more efficient than rate performance standards but are rarely used. Some point-source air and water regulations come close to the cap format by limiting the tons of BOD or other pollutants discharged per month or year. In practice, however, a larger limit is assigned to a larger plant, so the performance standard in effect is levied on the rate, *i.e.*, tons per unit of plant capacity. New discharge limits based on total maximum daily loads (see Section 4.5) may be closer to a true plant-level cap.

Non-economists often fail to appreciate the value of the flexibility provided by market instruments and, equivalently, the costs – often hidden – of the constraints imposed by non-market regulations.⁸ A large body of literature has attempted to estimate the cost-savings of using a market instrument for the regulation of a specific pollutant. These estimates have added empirical weight to the qualitative arguments of this section. The *2003 Economic Report of the President* reprints a figure from Field (1997) showing ratios of costs of command-and-control (*i.e.*, non-market) to least-cost (market-based) regulation that

⁶This argument relies on the quantity of pollution rather than its rate or concentration being the source of the environmental externality. In some cases, both quantities and concentrations cause negative externalities.

⁷This discussion does not address the further issue of emissions from new versus old cars.

⁸On the other hand, economists often fail to recognize the difficulty in monitoring emissions or other environmentally damaging behavior accurately and precisely, a feature that is necessary for market regulations to work effectively.

range from 1.07 (for sulfates) to 22 (particulates). Prominent studies of these cost savings include Atkinson and Lewis (1974); Atkinson and Tietenberg (1982); Seskin et al. (1983); and O’Ryan (1996).

The general lesson from the empirical literature is that the cost savings from market instruments can be dramatic. Studies typically estimate that control costs will be four times higher under the relevant non-market alternative than under cap-and-trade; in many instances, the ratio is even higher. Market mechanisms, where they can be used, always result in lower aggregate control costs than a comparable non-market one. The cost-savings that might be realized in any particular instance, however, are highly dependent on the nature of the non-market alternative, since “non-market” covers a wide range of alternatives with a wide range of efficiencies.

If anything, the savings from market approaches are apt to be *underestimated* for two reasons: (i) Savings estimates are based on currently available control strategies and usually do not take account of new strategies that are likely to be discovered; (ii) Market approaches are almost always less costly to administer. These cost savings are difficult to value but they reinforce the static cost-savings modeled above.

Taxes and cap-and-trade have further advantages beyond the cost-savings they provide. Tax rates and caps can be imposed with less technical knowledge about the regulated sector than required for technology or performance standards. Other regulations, which require greater technical knowledge of the regulated sector, must first be authorized by legislation and then the rules themselves issued subsequently by government agencies. The rulemaking process is subject to lawsuits and attendant uncertainty and delay in a way that legislation is not. This advantage of simplicity and transparency for market instruments often goes unappreciated.

The arguments of Sections 2.1 and 2.2 rely on two further conditions. First, the pollution or other environmentally-damaging behavior must be measurable. The item to be taxed, or for which permits are to be required, must be able to be observed by regulators with relative ease and substantial accuracy. The importance of this feature is clear but often unacknowledged. When the pollution (or its analog) cannot be measured, policymakers must choose between a market instrument

tailored to the “imperfect” item – that is, some action that is correlated with emissions or their analog – and a non-market approach, which lacks the advantages that this section has laid out.

Second, the action being targeted must be the source of the negative externality. The model of Section 2.1 shows the assumption clearly (that is, the regulation action, s , is the argument of the damage function), but many regulations do not target the true source of an externality. Again, it goes without saying that a regulation that is cost-effective but targeted at the wrong action is not necessarily cost-effective in curtailing the true source of the externality.

2.3 Other market-like mechanisms

2.3.1 Subsidies

Because a subsidy is a “negative tax,” it may, in some circumstances, lead to the same efficient outcome as a pollution tax. Those circumstances are almost never observed.

Subsidy can refer to many different types of policies, such as a subsidy to homeowners for installing insulation in their homes, a subsidy to the production of ethanol, or a subsidy to polluters directly on emissions reductions. The first type of subsidy is unlikely to be cost-effective because it exhibits very little within-source flexibility; only certain kinds of energy-efficiency investments are eligible for the subsidy. If these are not the lowest-cost activities then the subsidy has failed the basic criterion of cost-effectiveness. The second type of subsidy exhibits within-source flexibility, but ethanol is only one of many actions to reduce the perceived externality, so this subsidy may also fail to encourage lowest-cost actions.

The third type of subsidy, on emissions reductions, is formally equivalent to a negative pollution tax and therefore shares important efficiency properties with a tax. For early discussions on the symmetry of taxes and subsidies, see Kneese (1964); Bramhall and Mills (1966); and Kneese and Bower (1968). Emission subsidies could in principle be paid per unit of pollution reduction relative to some baseline amount of pollution. This set-up is equivalent to giving the firm a lump-sum payment and then taxing emissions. Subsidies of this sort have been

considered by economists but real-world applications have differed in key ways, so that they can rarely be considered a form of negative tax.

The formal symmetry between such a subsidy and a tax holds only if the baseline level of pollution is exogenous to the firm. If the baseline is not exogenous, then the firm has an incentive to undertake actions that increase its assigned baseline; these actions are inefficient Kamien et al. (1996).

When the baseline is exogenous to the firm and there is no dead-weight loss to taxation, a subsidy achieves the cost-effective pollution outcomes for a given set of firms. But we must also be concerned with the number of firms in the polluting industry. The long-run equilibrium will differ between a pollution tax and subsidy, since the latter involves a lump-sum payment which increases the incentive for the firm to stay in the industry (unless the payment is made even if the firm exits) or for other firms to enter the industry relative to a tax. Indeed, while the subsidy would lead to pollution reduction for each existing firm, it could lead to greater total pollution due to an increase in the number of firms in the industry (Baumol and Oates, 1988).⁹ To put this another way, the only truly exogenous baseline for plants that are not currently operating is zero.

In order to achieve long-term efficiency, the regulatory system would require that firms pay the total cost of their pollution (see Schulze and d'Arge, 1974; and Collinge and Oates, 1982). For example, a tax equal to marginal pollution damages but that varied the tax as marginal damage changed (due to changes in total pollution) would result in an efficient short-run and long-run outcome. The long-run efficiency would be achieved because each firm would end up paying the total external cost of its pollution. In this case, firms that contribute net benefits to society will remain in the industry, whereas those with negative net benefits will exit.

However, a system of varying taxes is difficult if not impossible to implement. If we instead consider a constant emissions tax and if marginal damages increase with emissions, then the constant emissions

⁹ Suppose a firm would earn negative profits under a pollution tax. It therefore would choose not to operate. Under a subsidy, the firm could earn positive profits, due to the lump-sum payment analogy.

tax will charge firms more than the total damages they cause. This would lead to an inefficiently high number of exiting from the industry.¹⁰ However, if we assume that marginal damages are relatively constant, at least over the range at which pollution is likely to vary due to entry and exit, then a constant emissions tax will result in firms paying both the marginal cost of damage at equilibrium and the total cost of the damage, resulting in an efficient short and long term outcome (see Schulze and d'Arge, 1974; and Spulber, 1985).

2.3.2 Other regulatory mechanisms

A few other regulatory approaches are sometimes labeled market-based. (Stavins, 2000, 2003) includes “market friction reductions” and “government subsidy reductions” as categories of market-based instruments. Market friction reductions include: (i) actions that create or enhance markets for inputs and outputs associated with environmental quality, such as recent laws that seek to increase competition in the electricity markets or laws that require that the government purchase recycled paper, thus enhancing the market for recyclables; (ii) liability rules that encourage firms to consider the costs of the environmental damages they produce; and (iii) information programs that aid market decision-making. Government subsidy reductions include reducing subsidies that promote environmentally harmful activities, such as subsidies that promote the use of fossil fuels (see, for example, Shelby et al., 1997).

EPA (2001) classifies regulatory approaches slightly differently and considers, in the context of “economic incentives,” additional policies (beyond taxes and cap-and-trade) that include: (iv) deposit-refund schemes; (v) liability; (vi) information disclosure; and (vii) voluntary programs.

Of these approaches, only deposit-refund and liability potentially satisfy the cost-effectiveness or efficiency criteria laid out by Baumol and Oates. Deposit-refund programs are economically equivalent to a tax or, more precisely, a tax-subsidy combination. They are suitable for such a small proportion of environmental problems (namely, those

¹⁰ Likewise, if marginal damages decrease with emissions, a constant emissions fee will lead to too much entry into the industry.

with an identifiable good that can be “returned” and that produces environmental damage only if it is not returned) that we do not cover them here. Liability is also economically equivalent to a tax, but its ability to achieve cost effectiveness or efficiency depends on the specific externality at stake and the workings of the court system.

2.4 Dissenting voices

There have been two prominent types of responses to these arguments of the superiority of market mechanisms. Some authors accept the general cost-effectiveness premise but argue that market power, localized concentrations of pollution (known as hot spots), or other conditions will lead market mechanisms not to reach their full potential in the majority of situations, and that these conditions do not afflict performance standards or other non-market regulations to the same degree. Sagoff (1997), for example, argues, in the context of an international cap-and-trade for greenhouse gases, that the initial assignment of permit rights is intractable; therefore, cap-and-trade for this problem is a “non-starter.”

A second, more prevalent argument departs from the economics paradigm to claim that cap-and-trade or taxes have other undesirable properties unrelated to specific market conditions. These claims question whether it is moral to allow the purchasing of the right to pollute. For example, Sandel (1997) writes, “Turning pollution into a commodity to be bought and sold removes the moral stigma that is properly assigned with it” and claims that trading may undermine “a sense of shared responsibility” in taking on environmental problems. Similarly, Goodin (1994) draws an analogy between environmental taxes and the selling of religious indulgences. He suggests that labeling environmental taxes as fines would be preferable, since fines provide exactly the same “balance sheet” disincentive but also bear a more explicit opprobrium.

Economists have been fairly dismissive of these anti-market claims. Moral stigma or a sense of shared responsibility is woefully inadequate to control environmental damage. Furthermore, whatever moral stigma may be lost from a market approach – and it is not at all clear that moral stigma or its effects are diminished by taxes

or cap-and-trade – must be balanced against the cost-savings, transparency, and reduced administrative costs that market approaches provide. Furthermore, lower-cost pollution control should allow society to be more vigorous in reducing environmental damage, not less.

Some claims against market approaches rely on the starker (and often unstated) belief that pollution control strategies should not depend on the costs. There is little that economists can say to counter such arguments except to show the consequences of ignoring costs.

There are also arguments against the use of market instruments as the sole regulatory approach. These are based on non-neoclassical behavior by consumers or firms. For example, if individuals are not forward-looking when they buy cars then mileage standards may be warranted in addition to a market mechanism that raises gas or electricity prices.

3

Issues in the Design of Market Instruments

The previous section presented the case for market mechanisms. This section examines the issues in policy design.

3.1 Cap-and-trade with banking and borrowing

Cap-and-trade reduces control costs by allowing both within-firm and across-firm flexibility in meeting the overall cap. Another potential source of cost reduction stems from allowing flexibility in emissions control across time. This intertemporal flexibility is achieved by allowing firms to bank unused permits for future use and to borrow future permits for use in the current period. These operations mean that annual aggregate emissions may deviate from the established annual cap in any particular year, although the multi-year sum of emissions will still equal the multi-year sum of the caps.

In order to prevent indefinite borrowing of future permits, which would effectively lead to a loosening of the cap for any fixed period of time, the regulatory authority could require that the cumulative deficit of permits be repaid by the end of a given planning horizon. An alternative method would be for the regulatory authority to levy a “tax” on any borrowed permits.

The ability to bank and borrow permits allows firms to shift their emissions across time. If the annual cap is becoming more restrictive over time and control costs are relatively constant, then firms will bank permits in the early years to be used later when the cap tightens, thus shifting their emissions later in time (Rubin, 1996). Emission shifting across time could also occur due to anticipated changes in the costs of abatement.

Under certainty, the market equilibrium price of a permit will rise at the rate of interest (Tietenberg, 1985; Kling and Rubin, 1997). This outcome makes intuitive sense, because if the price of permits increased faster than the rate of interest there would be an arbitrage opportunity to buy more permits earlier, bank them, and sell them later at a profit. If the price of permits rose at less than the rate of interest then a similar opportunity would present itself through borrowing. Cronshaw and Kruse (1996) show that permit prices should rise at the interest rate even if some (but not all) firms are subject to profit regulation. An implication of this result is that the spot price of allowances that cannot be used until a specified future year should be equal to the spot price of allowances that are eligible for current emissions, provided uncertainty is small (Bailey, 1998). Bailey (1998) shows that this property holds for sulfur dioxide permits issued under the Acid Rain Trading Program.

Other authors have focused on the welfare effects of banking and borrowing, which are affected by the control costs and by the intertemporal pattern of pollution. Kling and Rubin (1996) argue that banking and borrowing will almost surely be welfare-reducing because firms want to push control costs (and therefore pollution reduction) into the future, but society will want pollution to be roughly constant over time. Schennach (1998) presents a model in which plants want to bank permits but does not examine the social welfare implications of banking.

In general, banking and borrowing help reduce the overall costs of compliance, so the overall social welfare effects depend on the benefits of pollution reduction over time. For a flow pollutant, if marginal damages are constant or increasing in pollution, control costs are constant over time, and the cap is becoming more restrictive over time, then allowing banking would be welfare enhancing. These are roughly the conditions that characterize the Acid Rain Trading Program, which

did exhibit permit banking in the early years. Note, however, that this social welfare claim is due to the inefficiency (under these assumptions) of having the cap become tighter over time.

Because of uncertainty in real-world contexts, at least some banking or borrowing should be allowed in order for the market to clear efficiently at the end of the accounting period.

3.2 Prices versus quantities: The choice between taxes and cap-and-trade

Economists and practitioners who desire the merits of a market mechanism must choose between taxes and cap-and-trade. This section reviews the theory on that choice, often called “prices versus quantities.” Later sections examine this question from an applied point of view.

For any cap in a cap-and-trade program, there is a corresponding tax that yields the identical efficiency results (Baumol and Oates, 1988). The conditions that govern this correspondence can change, however. For example, if a given tax and cap would lead to the same amount of carbon emissions, a reduction in the costs of carbon abatement, such as an improvement in energy efficiency technology, would cause carbon emissions to decrease under the tax but not under the cap-and-trade. Since changes in costs, technologies, and other factors are inevitable, it is desirable to examine the two policies under different conditions of uncertainty about those future economic conditions.

3.2.1 Uncertainty and linearity

The question of taxes versus cap-and-trade was first addressed by Weitzman (1974). Weitzman showed that when the benefits of improved environmental quality are unknown but there is no uncertainty over pollution control costs, then taxes and cap-and-trade are equally efficient (see also Adar and Griffin, 1976). When the benefits of environmental quality are known but there is uncertainty about the costs of pollution control, then the two mechanisms diverge.¹ Weitzman derived the

¹Stavins (1996) notes several papers preceding Weitzman that also tackled this issue.

following expression for the relative advantage of taxes over cap-and-trade when the benefits of pollution reduction are known:

$$\Delta \approx \frac{\sigma^2 B}{2C^2} + \frac{\sigma^2}{2C}, \quad (3.1)$$

where B is the slope of the (linear) marginal benefit curve, C is the slope of the (linear) marginal cost curve, and σ^2 is the variance of the vertical shifts in the marginal cost curve. When $\Delta > 0$, taxes are superior to cap-and-trade; when $\Delta < 0$, cap-and-trade is superior. If the benefits of pollution reduction are strictly concave, as typically assumed, then $B < 0$, so Δ may take on either sign under normal assumptions.

This expression suggests that if the marginal benefits of pollution reduction are known to be relatively inelastic, then a quantity instrument (cap-and-trade) will have greater expected net benefits than a price instrument (tax). With a relatively steep marginal benefit curve, fixing the level of pollution reduction leads to levels of control that are closer to the optimal than a price control once the uncertainty is resolved. In contrast, if the marginal benefits of pollution reduction are known to be relatively elastic, then a price instrument will have greater expected net benefits than a quantity instrument because fixing the price leads to levels of control that are closer to the optimal than quantity control once the uncertainty is resolved.

The reason for the divergence of the implications of quantity and price instruments is that a static price instrument fixes the marginal cost of abatement across firms, so as the marginal abatement cost curves shift over time the level of pollution reduction varies. In contrast, a static quantity instrument fixes the level of pollution, yet as the marginal abatement cost curves shift over time, the level of marginal costs varies over time. For example, technological improvements that reduce control costs will lead to a decrease in total emissions under a price control, but will leave total emissions unchanged under a quantity control. Similarly, a pollution tax not tied to inflation will lead to an increase in emissions as overall prices rise, but such price increases will have no effect on total emissions under a quantity constraint. Without information about the benefit structure of pollution reduction over time, the choice of a quantity versus a price instrument does not have

any *a priori* effect on efficiency. Nonetheless, policymakers may place different emphasis on the desire to fix marginal costs with certainty versus the desire to fix emissions with certainty.

Weitzman's results apply only to the case where taxes are linear, an insight implied by Dasgupta et al. (1979). If taxes can be nonlinear and can depend on the aggregate environmental damage, then taxes are always superior to cap-and-trade. The reason is that the optimal tax is equal to the marginal social damage and is independent of control costs.² Thus, even if damages are uncertain, if the uncertainty can be reflected in a possibly nonlinear tax, taxes are always preferable to cap-and-trade. The optimal cap, in contrast, depends on both control costs and environmental damages. This additional dependence on control costs makes cap-and-trade inferior to fully general environmental taxes. A corollary is that when environmental damages are linear ($B = 0$), taxes are always superior to cap-and-trade.

Nonlinear taxes, whether environmental or non-environmental, are an interesting theoretical construct and have received a tremendous amount of attention from economists since the pioneering work of Mirrlees (1971), but nonlinear environmental taxes are difficult to imagine in practice. The tax modeled in Dasgupta et al. (1979) is a function of all sources' emissions; in practice, this makes it a function of aggregate environmental damage. Such a set-up means that a source would not know the exact tax it will pay until all emissions have been measured.

Note that if taxes can be nonlinear in a source's emissions but cannot depend on other sources' emissions or on aggregate environmental damages, then taxes lose their uniform superiority over cap-and-trade, but they remain superior over a wider range of parameters than implied by Equation (3.1).

In summary, environmental economists are not sacrificing much by focusing on linear taxes, so Weitzman's results remain instructive.

Subsequent authors have examined the Weitzman result under further uncertainty assumptions, though maintaining the assumption of linear taxes. Stavins (1996) conjectured that uncertain benefit and cost

²This result fails if there is a deadweight loss to taxation. See, for example, Spulber (1985).

parameters may be correlated and argued that a positive correlation tends to favor cap-and-trade. Karp and Hoel (2002) has examined the question when the pollutant is a stock pollutant. Numerous articles, including Weitzman (2002), have tackled the question for fisheries.

This literature remains somewhat unsatisfying, however, because it highlights the dependence of the optimal tax or cap-and-trade on the estimate of environmental damages, including its shape. In the 1970's, when Montgomery (1972); Baumol and Oates (1975); Weitzman (1974) and others were exploring cap-and-trade and environmental taxes, the science of environmental valuation was in its infancy. It was easier to imagine then that environmental damages would be able to be measured concretely. Indeed, the papers added to the rising support for environmental valuation since they showed the important role played by damage estimates in the question of optimal regulation. The Clean Air Act itself embodies this viewpoint that damages would be straightforward to assess since it places a great deal of weight on the ability of scientists to find a safe pollution level.

Subsequent events have shown us that damages are not easy to estimate and that those estimates that have been undertaken may be subject to skepticism. Therefore, it seems problematic to base environmental policy choices on very specific information as the level and shape of aggregate damages.³ This dilemma leaves us without an operational but still economics-based approach to the choice between taxes and cap-and-trade.

3.2.2 Incentives for investment and innovation

A second criterion used to inform the choice between taxes and cap-and-trade is their roles in fostering innovation in new control technologies or investment in (or adoption of) existing technologies. This literature lacks the unifying insight that Weitzman provided for the case of

³Baumol and Oates (1988), wise as usual, anticipated this course of events: "There is a promising body of work applying a variety of techniques to the valuation of the damages from a polluted environment. However, it is hard to be sanguine about the availability, in the foreseeable future, of a comprehensive body of statistics reporting the marginal net damage of the various externality-generating activities in the economy" (p. 160).

uncertainty (see, for example, Requate and Unold, 2003), but there are two broad conclusions.

First, taxes and cap-and-trade in general have similar effects on innovation and adoption of control technologies, and will often provide socially optimal levels of these activities (Downing and White, 1986; Carraro and Siniscalco, 1994). The key reason is both the within-firm and across-firm flexibility that foster such activities and the continuous incentives that market mechanisms provide for pollution reduction.

The *continuous-incentive* advantage of market mechanisms has not received as wide recognition as flexibility, although the intuition is straightforward. Under taxes and cap-and-trade, polluters always face an incentive to reduce their pollution, even as technologies evolve. There is no “cut-off” at which the incentive stops or lessens.⁴ Such a cut-off exists under most non-market regulations, leading to a reduction in incentives for pollution control once the cut-off is reached. It should be obvious that the continuous-incentive property matters in a dynamic sense, not a static sense, which is why we include it in this section.

Second, most of the predicted differences in the effects of taxes and cap-and-trade are due to assumptions about the ability of the regulator to observe and react to these activities by changing the tax or cap, rather than to inherent properties of taxes or cap-and-trade. The regulator’s ability to adjust the regulation is a key element of Downing and White (1986); Milliman and Prince (1989); Jung et al. (1996); and Requate and Unold (2003), among others.

As with the uncertainty case, a linear damage function is an important benchmark. Because the optimal tax depends only on the social damage and is independent of control costs, the regulator would not need to update the tax as new technologies lower the cost of pollution control. He would, however, want to update a cap. The necessity of adjusting the optimal cap may be disadvantageous either if the regulator is assumed not to be able to update or if the regulated sources can influence the cap through strategic investment that changes the

⁴The incentive for pollution reduction under cap-and-trade can stop if the cap is reached at no cost in a given period and there is no banking, or if the cap is reached at no cost permanently. However, most cap-and-trade proposals assume that the cap will be binding.

cost of pollution control, thus changing the optimal cap. Thus, the tax is again superior to cap-and-trade. This intuition underlies Biglaiser et al. (1995). This superiority result further depends on there being no opportunity cost to public funds, a case considered by Laffont and Tirole (1996). When damages are not linear but the tax is, the tax must be updated as costs change, just like the cap. Keohane (2003) suggests a two-part tariff in such a situation.

A substantial literature focuses not only on prices versus quantities but on broader questions, such as market versus non-market mechanisms. Maleug (1989) noted that pollution standards (that is, performance standards) may be superior to market mechanisms. Other parts of the literature have looked at the properties of the optimal tax, where optimality includes the costs of innovation or investment. Biglaiser et al. (1995) note that innovation is different from adoption (or investment) because of the public good properties of innovation.

3.3 Hybrid instruments: Cap-and-trade with a safety-valve price

The choice of a regulatory instrument is not restricted to either a price or a quantity control. As first demonstrated by Roberts and Spence (1976), it is possible to devise a mixed, or hybrid, instrument that sets a cap on emissions but allows a loosening of the cap at a fixed trigger price or “safety-valve.”

Under this system, the regulator commits to making as many permits available as demanded at an announced safety-valve price. This effectively sets an upper limit on the price of a permit, at the tradeoff of allowing the quantity cap to be violated if this price limit is reached; the safety-valve price then operates as a pollution tax. The purpose is to hedge the possibility of unexpectedly large control costs. Roberts and Spence (1976) demonstrate that, when the regulatory agency is uncertain about abatement costs, a hybrid approach can have lower expected costs than either a strict price or quantity instrument.

The reasoning is straightforward. For a given benefits schedule, if abatement costs turn out to be higher than expected, then a tax will bring about too little pollution abatement and a cap will bring about

too much pollution abatement. Conversely, if abatement costs turn out to be lower than expected, then a tax will bring about too much pollution abatement and a cap will bring about too little pollution abatement. By using a hybrid approach, the regulatory authority can establish a cap that applies only when control costs are low and a tax that only applies when control costs are high.⁵

In the case of climate change, Pizer (1998) estimated that the optimal hybrid approach leads to slightly higher net benefits than the optimal (linear) tax, and much larger welfare gains than the optimal cap-and-trade policy. Because climate models suggest relatively flat marginal benefits of regulating greenhouse gases, the result of the relative merit of a hybrid or tax approach over a cap stems directly from Equation (3.1).

Some of the U.S. policies that contain a safety valve also require that the stock of future permits be reduced one-for-one for any current permits sold at the safety valve price. The theoretical implications of this provision have not been examined in the literature.

3.4 Market imperfections

In discussing both the efficiency and the cost-effectiveness goals, we have thus far assumed competitive markets for output and for the pollution permits in the regulated markets. We next examine the impact of market power in the output and permit markets.

As shown by Buchanan (1967) a per unit tax set at the marginal external cost will not necessarily improve social welfare in a non-competitive market. This finding is straightforward: given that a monopoly produces less than a competitive market, it is possible that an unregulated monopoly already produces at or below the efficient amount even after accounting for externalities. Thus, a tax on such a firm would restrict production by even more and result in a welfare loss relative to not levying a tax. Lee (1975) shows that if different sources

⁵Conversely, as described by Roberts and Spence (1976), the regulatory authority could also set a per unit subsidy for over-compliance. This way, after establishing the quantity cap, if costs are lower than expected, there is some incentive for firms to reduce pollution by more than is necessary by the cap.

of an externality have different levels of market power, than a uniform per unit tax on emissions will not be efficient.

The problem with regulating an externality-producing non-competitive firm is complicated because there are two sources of inefficiency: one is the distortion caused by the externality, the other is the distortion caused by the imperfectly competitive market. A tax on emissions will reduce the distortion caused by the externality but increase the distortion associated with the inefficiently low output production of a non-competitive firm. Optimal policy would require a Pigouvian tax to be coupled with a per unit output subsidy (Barnett, 1980; Oates and Strassmann, 1984). More generally, the government may wish to address directly the source of non-competitive behavior in the product market and then deal with the environmental problem using the principles in Section 2. However, given that it is unlikely that governments can squelch all non-competitive behavior, a second-best alternative would be an emissions tax that optimizes across the two distortions.

Lee (1975) and Barnett (1980) both derive this second-best emissions tax, which equals the efficient Pigouvian tax on a competitive firm minus the ratio of marginal abatement cost for the firm divided by price elasticity of demand for the firm's output. The information constraints on deriving this second-best emissions fee are quite burdensome since they require the regulatory agency to know the price elasticity of demand and the marginal abatement cost for each firm. Nonetheless, Oates and Strassmann (1984) demonstrate that under certain conditions a uniform emissions fee that ignores the market imperfections results in welfare gains from pollution reduction that outweigh the welfare losses caused by exacerbating the low production due to market power.

Even when output markets are perfectly competitive, the permit market under cap-and-trade may not be. A firm with market power in the permit market can influence the permit price such that all firms do not face the same permit price and therefore do not have equal marginal abatement costs, a requirement for cost-effectiveness. Hahn (1984) shows that the distribution of permits in a competitive market does not affect aggregate abatement costs, but may affect aggregate costs in a non-competitive market. For example, when one firm has

permit-market power, the initial permit distribution is predicted to lead to a cost-effective outcome only if the firm with market power is allocated its equilibrium number of permits. When the permit market is likely to be non-competitive, an environmental tax may be superior to cap-and-trade. For recent work on market power in a permit market that allows banking, see (Liski and Montero, 2005a,b).

A related literature examines the choice of taxes versus cap-and-trade when there is incomplete enforcement. Viscusi and Zeckhauser (1979) assessed the impact of enforcement on taxes versus standards. Malik (1990) looked at the role of incomplete enforcement in the context of cap-and-trade. In a recent study, Montero (2002) finds that a second-best outcome can be achieved equally well with either the tax or cap-and-trade instrument, so long as the benefits and costs are known with certainty. In the uncertain case, the quantity instrument performs better.

3.5 Permit allocation, revenue recycling, and the tax-interaction effect

Another distinction between an environmental tax and cap-and-trade is that the tax always generates public revenue. A cap-and-trade system generates public revenue only when at least some of the permits are auctioned off by the government. In practice, cap-and-trade systems have almost always distributed permits to existing regulated entities without charge, a procedure known as grandfathering, which generates no revenue for the public coffers.

Grandfathering rules should be based on polluter characteristics that preceded the announcement of the cap-and-trade. Otherwise, regulated sources will have incentives to undertake actions to increase their share of the permits, which is inefficient. Grandfathering rules, even those that are contemporaneously exogenous, may also lead to rent-seeking over the design of the rule, a further source of efficiency loss.

The decision between permit auctioning (or a tax) and permit grandfathering is believed to have efficiency consequences even when the permit market is competitive and the grandfathering rule does not affect firm behavior. Because environmental regulation, by whatever

means, raises output prices (through increases in the price of electricity or transportation, for example), it can magnify distortions from other taxes such as income or sales taxes (Sandmo, 1975; Parry, 1995, 1997; Bovenberg and Goulder, 1996; and Goulder, 1998.)⁶ This *tax interaction effect* reduces the efficiency of the regulation. Goulder (1998) showed that it is possible that the tax-interaction effect will actually lead to negative net benefits for environmental policies; this could occur if those policies had relatively small marginal benefits.

A pollution tax or cap-and-trade with auctioned permits generates government revenue that can be used to offset some of the distortionary taxes. This revenue can be used to reduce those distortionary taxes, thus reducing public finance inefficiency (Terkla, 1984; Lee and Misiolek, 1986; Goulder, 1998); this is the so-called *double dividend effect* of environmental taxes. The ability to use environmental tax or permit revenue to reduce other taxes is known as the *revenue-recycling effect*. A cap-and-trade system that grandfatheres the permits fails to take advantage of the double dividend effect.

A further problem in the initial allocation of permits is the decision over which polluters will be covered by the cap-and-trade system. Under taxes or auctioned permits, this decision is straightforward: the policy should cover as many polluters as possible, except when the costs of pollution monitoring are high or, in the extreme, pollution monitoring is impossible for a class of firms.

3.6 Opt-in

In the case of cap-and-trade, a regulator may allow sources not originally covered by the regulation to (voluntarily) *opt in* to the system. Sources would do this if they expect to receive an allocation of permits that exceeds their expected emissions, thus making them net sellers of permits.⁷ Sources might also opt in if doing so allowed them to bypass

⁶ Roughly speaking, the tax distortion is magnified if the rise in commodity prices due to environmental regulation leads to a reduction in the supply of labor. This effect would occur if those goods are substitutes for leisure.

⁷ More precisely, a profit maximizing firm will opt in if the expected present value benefit of their permit allocation exceeds the expected present value cost of any subsequent emission reductions.

other regulations. The opt-in issue currently arises for non-point water pollution, greenhouse gas emissions (both point and non-point sources), and some air pollution point sources. The perceived advantage of opt-ins is that additional sources bring additional abatement options, which lower total pollution control costs by increasing across-firm flexibility.

The welfare implications of an opt-in provision depend critically on the rules for permit allocation for sources that choose to opt-in. As discussed in the previous sub-section, absent an auction system, permit allocation should be based on polluter characteristics that precede the announcement of the cap-and-trade. Otherwise, the opt-in sources will have incentives to game the system, thus reducing social welfare. Social welfare will also be reduced if the opt-in provision restricts the number of permits a source can sell for a given control action. For example, a greenhouse gas cap-and-trade system that only gives partial credit for sequestration is not efficient.

The welfare implications of an opt-in provision also depend on the number of permits allocated to the sources. If the allocation is below the source's projected emissions, typically referred to as its business-as-usual baseline, then the source lacks incentive to opt in even if it has socially valuable control options. Note that the source might still opt in if the cost of reducing emissions below the permit level is offset by the revenue gained from selling the surplus permits. If the source does not opt in, then social welfare is unchanged. If the source does opt in, then social welfare is increased because the reduction in the aggregate emissions cap is achieved at a lower total cost. On the other hand, if the allocation is above the source's projected emissions, then the source has incentive to opt in but doing so raises the effective cap on emissions. Welfare may or may not be increased in such a situation.

These problems are exacerbated because the voluntary nature of the opt-in provision presents a selection problem: Sources that receive an allocation below expected emissions (in which case the welfare implications are non-negative) have less incentive to opt in than sources that receive an allocation above expected emissions (in which case there are ambiguous welfare implications). Thus, the key to the opt-in problem is determining the baselines of interested unregulated sources. This problem is made even more complicated by the possibility that

(i) the business-as-usual emissions path for the unregulated source is not constant but varies from year to year, and (ii) the business-as-usual emissions will be affected by the policy or by other sources' opt-in decisions.

In all cases, once a source has opted in, it should be treated as fully regulated source for all future years. Any subsequent increase in its emissions, regardless of its *ex ante* baseline, must be covered by the tax or cap-and-trade. The welfare gains from temporary opt-ins are likely to be quite small or even negative.

One final point is worth reiterating. Opt-in is fully efficient only in rare circumstances, namely the case in which the permit allocation is exactly equal to business-as-usual. To the extent possible, all sources should be covered from the start by the tax or cap-and-trade, rather than mandating regulation of some sources while allowing others to opt in.

3.7 Spatial dimensions

As discussed in Section 2, efficient policy results from setting the price of pollution at the level of marginal external damages. For most pollutants, this goal is difficult to achieve because the effects of emissions depend on where the emissions occur. Pollution that primarily affects human health will be more damaging if it is upwind of a large city than if it is upwind of a small city. Water pollution in the vicinity of a bountiful fishery will have greater effects than water pollution of a less productive water body. Some forms of water pollution degrade over distance; others have their primary negative effects when they reach a stagnant body of water. Indeed, in very few cases will damages truly be independent of where the pollution is emitted. The only example we can think of is greenhouse gases, whose contribution to climate change is independent of where the gas is emitted.

A market-based policy that sets a uniform price for pollution and thus ignores the differential spatial impacts could be inefficient if it leads to high concentrations of the pollutant in a localized area. These occurrences, known as "hot spots," could result in a substantial

increase in damages if the marginal damages of the pollutant increase dramatically with concentration and if the hot spot occurs in a large and highly-exposed population. In general, the policy design solution is for the cap-and-trade system to include equivalence ratios – essentially exchange rates – that reflect the relative externality due to the location of the source of the emissions. The tax system should vary the tax in a similar way (Baumol and Oates, 1988). Nonetheless, equivalence ratios and varying taxes have rarely been included in actual cap-and-trade and tax designs.

3.8 Non-point sources

Taxes and cap-and-trade both assume that the pollutant of concern can be measured and monitored. Many environmental externalities, such as agricultural runoff of chemicals, sediment, and nutrients, are known as non-point externalities, since they cannot be measured through direct monitoring of sources. Without direct monitoring, emission fees and cap-and-trade cannot be assessed or enforced. Any attempt to adapt effectively these regulatory instruments to non-point externalities requires information on the relationship between production inputs and the environmental externality (Meade, 1952; Griffin and Bromley, 1982). With direct understanding of this relationship, the least-cost option would be to create implicit prices on the inputs or management activities that lead to the agricultural spill-off (Griffin and Bromley, 1982; Shortle and Dunn, 1986).

If the pollution function varies across sources, then cost-effective achievement of the emission goal would entail non-uniform instruments. Given that it is unlikely that a regulatory agency can charge each input source a different price, Helfand and House (1995) estimate whether uniform input taxes offer a second-best instrument. They find that the uniform instruments may not be costly relative to a first-best instrument.

Where the pollution function is unknown (or can only be known at great costs), Segerson (1988) suggests an incentive instrument based on ambient concentrations of the pollutant. Each source is subject to

a tax (subsidy) payment for each unit of total ambient levels above (below) the pre-determined goal. There is also a penalty imposed on each firm if ambient levels exceed the pre-determined goal. While this incentive system can be cost-effective, it requires setting different tax and subsidy levels across firms, which is difficult to implement.

4

Policies in Practice

In 1989, Robert Hahn published the influential article, “Economic Prescriptions for Environmental Problems: How the Patient Followed the Doctor’s Orders,” which examined early experiences, in the U.S. and elsewhere, with both cap-and-trade and environmental taxes.¹ In the years since, market-based policies have flourished. They have gone from a half-dozen limited applications in air and water pollution to hundreds, enacted or proposed, targeted at nearly every form of environmental problem. The most noteworthy U.S. market-based policy, the Acid Rain Trading Program, was only in the blueprint stage at the time of Hahn’s article, and this influential policy now serves as the model for yet grander legislative cap-and-trade proposals such as the Clear Skies Act and the Climate Stewardship Act. Given the dramatic increase in the number of market-based environmental policies, economists can evaluate market-based approaches with a depth that was unavailable to Hahn in 1989.

The U.S. experience in the intervening years allows us to say much more about the performance of market mechanisms, in a much wider variety of situations. This expanded application continues to be

¹For a more recent treatment of the political economy of this issue, see Keohane et al. (1998). Berthold (1994) also provides useful insights on this question.

accompanied by many non-market elements. Even in the best of the cap-and-trade applications, non-market elements have been appended to the basic cap-and-trade format. This section highlights these mixed approaches.

At least two additional issues have come to light as the use of market-based mechanisms has expanded, namely the *misplaced cap* problem, which occurs when the market instrument does not directly address the externality that it appears to be aimed at, and the *opt-in* problem, which arises when unregulated sources are brought into an existing cap-and-trade system.

Hahn (1989) suggested that environmental taxes were more prevalent than marketable permits. In part, this conclusion was the result of his including European policies in his assessment, which appeared at that time to rely more often on taxes. Our appraisal shows a greater reliance on marketable permits, which have clearly overtaken taxes as the regulatory instrument of choice. Indeed, cap-and-trade's prevalence appears to be increasingly true even in Europe, as demonstrated by the E.U.'s adoption of a cap-and-trade system rather than carbon taxes to regulate greenhouse gas emissions. In the few instances where the U.S. has adopted environmental taxes, they are primarily used as public finance tools rather than regulatory instruments. The consequence of this public-finance motivation is that tax rates tend to be small and to be accompanied by rather heavy-handed non-market regulations. As a result, we know much less about the real-world performance of environmental taxes.

In what follows, we describe in detail fourteen U.S. examples of market-based approaches. Each of these presents a different manifestation of market rules and exploitation of market principles. In each case, we examine the economics of the regulations with particular attention to the interaction of their market- and non-market-based provisions.

4.1 Emissions trading under the Clean Air Act

4.1.1 Clean Air Act overview

The Clean Air Act (CAA) arguably has the largest regulatory scope of any environmental statute. It also offers examples of the mixed

approach to pollution regulation since it includes both market and non-market instruments. This mixture has changed over time as the statute has been re-authorized and the regulatory framework developed.

At the core of the Clean Air Act are the National Ambient Air Quality Standards (NAAQSs), which are nationwide standards for six common pollutants that apply to all air quality regions within the U.S. The Act requires that the primary standards be set to protect the public health with an “adequate margin of safety,” without any consideration of cost. Regions that do not meet all of the standards are designated as nonattainment areas. A state with a nonattainment region must develop a state implementation plan (SIP), which is subject to EPA approval and which describes the mechanisms the state will use to address the nonattainment designations.

While the primary responsibility for meeting the air quality standards falls on the states, the Act does set out some nationwide requirements for new or modified sources rather than existing sources. All new or modified stationary sources must meet new source performance standards (NSPSs), which are command-and-control, technology-based standards set on the basis of the “best technological system of continuous emission reduction.”

More stringent regulatory constraints are placed on “major” new or modified stationary sources in nonattainment areas. In order to obtain a preconstruction permit, these sources must meet a uniform emission rate known as the lowest achievable emission rate (LAER). This standard is defined as the lowest emission rate for a class or category of a source in any state’s implementation plan, whether or not any source is currently achieving that emission rate. In order to obtain a permit, these newly constructed or modified sources must also offset their projected emissions by obtaining reductions from other sources in the nonattainment area.

State implementation plans must also contain measures necessary to prevent the degradation of air quality in areas that are in attainment of the standards, including a requirement that major new or modified sources use best available control technology (BACT), which are determined by states on a case-by-case basis. In principle, the NSPSs are supposed to serve as a minimal standard, with BACT and LAER

offering stricter controls. In practice, the NSPS is often considered satisfactory to meet the BACT or LAER standards (Portney, 2000).

4.1.2 Emissions trading

LAER, BACT, and NSPS standards are highly prescriptive and are often treated as archetypal examples of command-and-control regulations. Although the standards take the form of performance standards, which allow plants flexibility in meeting emission rate goals, in practice they require the use of specific control practices. However, some minor market approaches were introduced in the mid-1970s in an attempt to reduce the cost of meeting the CAA goals. One example is the offset program, which was mentioned earlier. This is the requirement that, in addition to meeting LAER, new or modified major sources must also offset their emissions with cutbacks in emissions from existing sources in the nonattainment area.² These new and modified sources effectively buy reductions from other sources in order to prevent a net addition in emissions in the nonattainment area.

In addition, a source can reduce its emissions beyond what is required by its legal obligations and then apply to the state for an emissions reduction credit. These credits can be used by the firm to offset an emissions increase in a nonattainment area, or they can be sold to another new or modifying firm in need of an offset in order to build in a nonattainment area, or they can be banked for later use. They can also be used in the bubble or netting programs introduced by the EPA.

The bubble program allows multiple emission sources in either attainment or nonattainment areas to be treated as one regulated source. This allows some within-plant flexibility in meeting emissions limits. One source within a bubble can under-comply with an emissions limit, given that another over-complies with to make up the difference. Similarly, a source within the bubble can under-comply and make up the difference by using an emissions reduction credit.

²Sources must offset more than their emissions. The exact offset ratio is determined by the state. Under some circumstances, sources can also offset their emissions from a source in another nonattainment area.

The netting program is similar to the bubble program except it specifically applies to the determination of whether a new or modifying source is even subject to the LAER, offsetting, and BACT requirements. Only “major” sources (those that emit more than 100 tons per year of a pollutant) are subject to these requirements. Before construction of a new or modified plant begins, netting allows sources to use emission reduction credits to offset the expected increase in emissions and thus not be characterized as a “major” source. That is, if a new or modifying plant’s expected emissions less the credits are less than 100 tons per year, then the plant is not considered a “major” source and thus is not subject to LAER or BACT standards (although they still must meet the NSPS requirement).

The commonly accepted verdict on banking, netting, bubbles, and offsets is that they added flexibility that reduced the cost of air quality regulation but have fallen far short of their potential (Tietenberg, 1985; Hahn, 1989; Hahn and Hester, 1989). For example, Hahn and Hester (1989) estimate that the emissions trading program has saved billions of dollars and has not inhibited the pollution reduction effectiveness of the Clean Air Act. However, they find that most of the cost savings have occurred due to within-firm flexibility. Very little cost-reducing trades have occurred across firms. They also find that banking is almost nonexistent.

The reason that cost savings have fallen short of the cost-effective outcome is largely because of the complex design of the trading program. Rather than assign sources pollution allowances which can be banked or traded, the emissions trading program assigns credits for *reductions*. In order to measure reductions accurately, the regulatory authority needs to know what emissions would be with business as usual. This is difficult to know, so the regulatory authority typically indirectly calculates the business-as-usual emission level by making inferences based on the control technology used by the source. Given the imprecision of this process, firms face great uncertainty about the number of credits they can expect given a certain pollution reduction plan.

The result is a complicated, uncertain, and costly process of achieving approval for credits (especially for credits traded across firms) from the regulatory authority. Thus, while the grafting of some

market-based flexible approaches onto the command-and-control structure of the Clean Air Act has reduced costs, the limitations of this approach relative to a pure cap-and-trade approach have restricted its potential effectiveness.

4.2 The Acid Rain trading program

4.2.1 Description

Perhaps the most notable example of a cap-and-trade system is the Acid Rain Trading Program, which was created as part of the Clean Air Act Amendments of 1990. The 1990 Amendments created a separate regulatory instrument for sulfur dioxide emissions from electric utility power plants without substantially altering the command-and-control requirements and rudimentary emissions trading discussed previously.

The statute established caps on these emissions that would reduce emissions by 10 million tons from the 1990 level by 2010, to be achieved in two phases. The first phase, which began in 1995, capped emissions at the largest emitting 263 units that were owned by 110 electric utility power plants located in 21 Eastern and Midwestern states. Another 182 units voluntarily opted into the first phase between 1995 and 2000. The cap was set based on an average emission rate of 2.5 pounds of sulfur dioxide per million BTU of heat input. The second phase, which began in 2000, further decreased the annual emissions of sulfur dioxide by establishing a cap based on an average emission rate of 1.2 pounds of sulfur dioxide per million BTU of heat input. This second phase cap was equivalent to 8.95 million tons and required all large fossil fuel-fired power plants (not just the initial 263) in the contiguous 48 states and the District of Columbia to hold allowances to cover their emissions.

This cap-and-trade program incorporates many features of a pure market-based approach. It allows within-plant flexibility, since the only effective obligation for the sources is that they submit allowances sufficient to cover their sulfur dioxide emissions. If a plant's emissions exceed its allowances, the firm is fined USD2,000 per ton of emissions that exceed the allowances and required to offset the excess amount the following year.

The Acid Rain Trading Program also allows across-plant flexibility. This flexibility is valuable because sources have multiple options for controlling emissions and these options differ across sources in their cost and suitability. Aside from reducing power generation, the two ways to reduce sulfur dioxide emissions are by switching to a fuel with lower sulfur content or by installing scrubber technology to reduce the sulfur dioxide exiting a stack. The cost of switching fuel versus installing scrubbers to achieve a given emissions reduction varies depending on specific market conditions and on characteristics of the source. By allowing sources flexibility in choosing their compliance strategies rather than establishing new emission rate limits or technology standards, the trading program reduces the overall compliance cost of the program. Ellerman et al., (1997) found that in the first year of the program, the reduction in emissions was achieved about equally by installing scrubbers and by switching to low-sulfur coal. By 2001, about 37 percent of reductions were due to scrubbers (Ellerman, 2003a).

Carlson et al. (2000) and Burtraw (1996) claim that the beginning of the cap-and-trade program saw cost reductions almost exclusively due to within-firm compliance flexibility. There were few trades conducted across firms, a finding that was likely due to the initial uncertainty of whether the allowance market would be well functioning. Thus, the cost savings from trading were initially relatively small. Carlson et al. (2000) estimate that this reluctance to trade led to control costs that were USD280 million and USD339 million (1995 dollars) higher than the least-cost solution in 1995 and 1996, respectively.

Other studies claimed that cost reductions from trading were further limited because public utility commissions established policies that gave utilities incentives to install scrubbers rather than purchase allowances (Rose, 2000). Other studies also suggested that trading in the early years did result in cost savings but the full cost-saving potential was not realized (Joskow et al., 1998; and Schmalensee et al., 1998).

It is also clear that the market has become more robust over time. Ellerman (2003b) finds that allowance prices available through various brokers and through EPA's auction (discussed later) seem to follow the

law of one price and that by 2001 about 12 million allowances had been transferred between economically distinct organizations. Additionally, there is a large amount of heterogeneity of emission rates across sources, with some above and some below the emission rate that would need to be achieved to meet the overall cap in the absence of trading. This rate heterogeneity is indirect evidence that trading is valuable (Ellerman et al., 1997; Ellerman, 2003b).

The Program also allows sources to bank unused allowances for future use. During the first phase of the program, regulated sources took advantage of this feature by banking allowances to use during the more restrictive second phase (Ellerman et al., 1997; Ellerman, 2003a). By the end of the first phase, sources had banked 11.65 million tons (Ellerman, 2003a). This banking of allowances in the first phase to be used in the more-restrictive second stage is consistent with optimal behavior suggested by theoretical models in Rubin (1996) and Ellerman and Montero (2002). The program does not, however, allow firms to borrow future allowances for current use.

Starting in 2000, the program set aside a very small number of allowances (50,000), which were available to be purchased at USD1,500 each, adjusted annually for inflation. This acts as a limited “safety valve” feature, although it has not yet been operable because the safety valve price is higher than observed allowance prices have been on the open market.

Another feature of the program is that it sets aside 2.8 percent of allowances each year that are then auctioned off in March. There are actually auctions for two types of allowances: one for allowances that can be used immediately and another for allowances that can only be used no earlier than the seventh year after they are offered for sale. In both cases, the auction revenue is transferred back pro rata to the sources from whom the auction pool was created.

The program allocates allowances each year to plants for free based on the plant’s baseline heat input characteristics, which were determined from historic production data. New sources are not allocated any allowances and must therefore purchase all their allowances on the open market or from the EPA’s auction.

4.2.2 Assessment

The Acid Rain Trading Program comes remarkably close to the pure cap-and-trade program that economists have long advocated. It does, however, contain some non-market elements.

First, the statute allocated “bonus” allowances to firms that installed scrubbers instead of achieving the reductions through other means. Second, new and modifying sources must still go through New Source Review, in which they demonstrate to regulators that their emissions and control strategies for sulfur dioxide and other regulated pollutants will satisfy the NSPS standards and will not increase emissions in nonattainment areas. NSR has been a source of considerable dissatisfaction by regulated plants and by economists who feel that new-source standards are costly and inefficient. It is unclear, however, how large a burden is imposed by NSR requirements for sulfur dioxide for utilities regulated under Title IV. It is widely believed that NSPS – which are clearly inefficient in the context of the cap-and-trade program – are not binding for sources covered by the Acid Rain Trading program; if so, they would not actually detract from the program’s true market character and resultant efficiency. Nevertheless, there is some evidence that the NSPS are at times binding and potentially costly. It is likewise unclear how large a burden is imposed by the performance standards component of New Source Review. This issue has not received as much attention from economists as it appears to deserve, given the prominence of the example set by the Acid Rain Trading program.

Another non-market feature is due not to the federal statute but to state level public utility commissions, many of which have adopted pricing rules that give utilities an incentive to install control technologies rather than, for example, switch coal types, reduce output, or purchase allowances.

The Acid Rain Trading Program exhibits at least one feature whose role was possibly unanticipated, namely the ability and willingness of citizens to purchase allowances and then retire them. Many of these purchases are by school groups, from elementary school to law school, and many are made at the yearly auction. The number of such allowances retired in this way has so far been small.

The Acid Rain Trading Program has a few failings not related to its non-market components but to broader design issues in the market mechanism itself. First, the program allows emissions to be freely traded across regions. This is inefficient because the environmental impacts of sulfur dioxide emissions vary by location. An efficient trading program would include a transfer coefficient that requires high impact polluters to hold relatively more allowances per unit of emissions. This inefficiency is not unique to the trading program, as the foundation of the Clean Air Act itself is a set of uniform, nationwide ambient standards, even though costs and benefits vary geographically. As we will discuss later, the recent Clean Air Interstate Rule does establish a transfer coefficient for trading across regions.

Second, because most allowances are distributed for free, and even those that are auctioned yield revenue that is redistributed in lump sum back to the regulated firms, the trading program fails to exploit the revenue-recycling effect. Goulder et al. (1997) estimate that the annual total cost of achieving a 10 million ton reduction in sulfur dioxide is 32 percent higher when the allowances are given away rather than auctioned off and the resulting revenue used to offset distortionary taxes.

A third question that must be asked of all cap-and-trade systems is whether the cap is set at the “right” level. Burtraw et al. (1997) argue that the prospective benefits of the sulfur dioxide cap exceeded the prospective costs, but this is not as strong as a claim about whether those net benefits were as large as possible, roughly speaking. Note that if the cap were set according to a Samuelsonian public good rule then the observed retirement of allowances would indicate that the cap was set too high.

4.3 The Clean Air Interstate Rule

Since the inception of the Clean Air Act in 1970, much of the regulatory focus has been on having states achieve attainment within all their regions. However, a provision within the Clean Air Act requires that state implementation plans “contain adequate provisions prohibiting . . . emissions activity within the state from emitting

any air pollutant in amounts which will contribute significantly to nonattainment . . . in any other state . . .” (section 110(a)(2)(D)). In 2005, the EPA found that 23 eastern states and the District of Columbia “contribute significantly” to nonattainment of the fine particle standard in other states, and that 25 eastern states and the District of Columbia contribute to nonattainment of the ozone standard in other states.

The EPA thus has statutory authority to require emission reductions of sulfur dioxide and nitrogen oxides in the violating states. Rather than require each state to arrive at its own individual plan to reduce emissions, the EPA offered the Clean Air Interstate Rule, which establishes regional cap-and-trade programs for both sulfur dioxide and nitrogen oxides for the regulated states. States are effectively assigned a budget of allowable emissions and then have the option of devising within their implementation plans means of reaching these budgets or of taking the expedited route of agreeing to participate in EPA’s regional trading programs. A state that must control both sulfur dioxide and nitrogen oxides can opt to participate in one of the trading programs and address the other pollutant through other means within its implementation plan.

The sulfur dioxide trading rule relies heavily on the existing Acid Rain Trading Program. The allocation of allowances to utilities remains unchanged from the original program. The goal of the new rule is to reduce sulfur dioxide emissions through two phases, the first starting in 2010 and the second reducing the cap further in 2015. Rather than reduce the allowance allocations, the more restrictive caps of the new trading program are met by changing the trade-in value of the allowances over time for sources within the regulated states. Whereas allowances issued under the Acid Rain Trading Program today can each offset 1 ton of sulfur dioxide emissions by a source, under the new rule any allowances issues between 2010 and 2014 will only offset 0.5 ton of emissions for a source located within a state regulated under this rule. Allowances issued on or after 2015 will only offset 0.35 ton of emissions for a source located within a state regulated under this rule. Aside from the change in the offset value of each allowance, the program retains all of the features of the existing Acid Rain Trading Program.

Grafting the new sulfur dioxide trading program onto the existing trading program does create some peculiarities. For example, the new program allows the banking of allowances, yet the value of the banked allowance is determined by the year it was issued, not the year it is expired. So a source can bank a vintage 2009 allowance and then use it to offset one ton of emissions in 2015; whereas a vintage 2015 allowance will only offset 0.35 ton of emissions in 2015. This creates an incentive for firms to bank allowances early on, before the change in the offset rate.

The program also creates a transfer coefficient for the value of allowances across states. For example, a vintage 2015 allowance will offset 0.35 ton of emissions for a source located in a state that is part of the regional trading program, but it will offset one ton of emissions for a source located in a state that is not part of the regional trading rule. This effect will likely be limited because most of the larger utilities are located within the eastern states that fall within the trading rule.

The new rule also establishes a nitrogen oxides trading program. In this program, the EPA determines the states' allowable emission levels for utilities, and then allocates the associated number of allowances to the states. States then have discretion to allocate the allowances to sources as they please. States can therefore opt for an auction approach should they desire. The nitrogen oxides reductions also occur in two phases. The first phase starts in 2009 and caps emissions at 1.5 million tons. The second phase starts in 2015 and caps emissions at 1.3 million tons. Utilities are allowed to bank any unused allowances for later use.

The main complication with the nitrogen oxides trading rule is that there was a pre-existing trading program for summertime emissions of nitrogen oxides (known as the NO_x SIP Call) for a subset of the states that are subject to the Clean Air Interstate Rule. As part of the Clean Air Interstate Rule, the EPA therefore designed two separate nitrogen oxides trading rules. States that were found to significantly contribute to nonattainment of fine particle standards in other states are subject to an annual cap. States that were found to significantly contribute to nonattainment of ozone standards in other states are subject

to an ozone season cap. Some states are subject to both caps. The ozone-season nitrogen oxides rule effectively replaces the pre-existing NO_x SIP Call. The new ozone-season program will be the only ozone-season NO_x program that EPA will administer; however, pre-2009 NO_x SIP Call allowances can be banked for later use in the ozone-season rule.³

The Clean Air Interstate Rule is clearly a result of the success of the original acid rain program. The real creativity behind the rule (first used in the NO_x SIP Call and upheld in the courts), is the way it crafts a market-based regulation onto the command-and-control laden Clean Air Act. Recall that the Clean Air Act requires that each state devise its own implementation plans for achieving attainment (subject to approval by the EPA). This makes it difficult to design a robust allowance market. By law, the Clean Air Interstate Rule must still allow states to devise their own regulatory plans, but it also offers an option for states to meet their implementation plan requirements by participating in the regional trading programs. The structure of these programs closely parallels the existing structure for the Acid Rain Trading Program, allowing within- and across-firm flexibility in meeting the new cap standards.

4.4 The Clean Air Mercury Rule

The NO_x SIP Call and the Clean Air Interstate Rule established the precedent of trying to fit cap-and-trade programs for electric utilities in the mold of the Acid Rain Trading Program within the existing command-and-control oriented Clean Air Act. Both of these programs fall within the Act's requirement for states to devise implementation plans to achieve attainment and to not significantly contribute to

³ An additional complication is that the pre-existing NO_x SIP Call regulated utilities and industrial boilers, whereas the Clean Air Interstate rule assigns emission budgets to states based only on utilities. Because the new ozone-season NO_x program will effectively replace the old NO_x SIP Call, this would leave the non-utilities with a limited number of trading partners. To address this concern, EPA allowed states to include the non-utility sources in the new ozone-season program. Yet another complication is that Rhode Island was subject to the NO_x SIP Call, but not the new trading rule. EPA allowed Rhode Island to meet its NO_x SIP Call obligation by participating in the new ozone-season program or to develop an alternative method for obtaining the required reductions.

nonattainment in other states. Neither of these plans can exempt new and modifying sources from the LAER, BACT, or NSPS standards where they apply. Therefore, those rules do not use cap-and-trade as the sole instrument for pollution regulation. But they both also establish regional cap-and-trade systems that achieve substantial reductions in emissions at costs lower than would be achieved through non-market approaches.

The 1990 Amendments to the Clean Air Act included a section on the regulation of a number of hazardous air pollutants, which are more acutely toxic, though typically more localized, than the criteria pollutants that are the focus of attainment standards. As part of this section, the EPA was required to specifically study the emissions of these hazardous air pollutants from electric utilities to determine if it was “appropriate and necessary” to regulate these sources under a stringent command-and-control system.

After EPA missed the deadline for this study, a lawsuit ensued that ultimately led to an “appropriate and necessary” finding on December 15, 2000 for the regulation of mercury (a hazardous air pollutant) emitted by electric utilities. This finding set the ball rolling for the establishment of a command-and-control standard known as Maximum Achievable Control Technology (MACT). MACT sets an emission rate standard equal to “the average emission limitation achieved by the best performing 12 percent of the existing sources . . . (CAA, Section 112(d)(3)(A)).” MACT sets an even more restrictive emission rate for new sources, which must meet “the emission control that is achieved in practice by the best controlled [existing] similar source . . . (CAA, Section 112(d)(3)).”

While the language that establishes these emission rates seems to suggest a precise requirement, there is some uncertainty and variability in what the MACT standard entails. For example, separate standards can be established for different categories or subcategories, with EPA having discretion on how to define these. Also, given the paucity of emission rate data for mercury from utilities, the EPA can allow for variability in the emission data, which can thus influence the determination of the specific emission rate standard. These reasons help to explain why some advocates of tighter mercury regulation claim

that the MACT should lead to 90 percent reduction of emissions, whereas the original MACT proposed (but never finalized) by EPA in 2004 would have led to approximately a 30 percent reduction in emissions.

The decision by the outgoing Clinton administration to regulate mercury from utilities using MACT seemed to have left the incoming Bush administration with the job of proposing and finalizing a command-and-control rule. However, in order to allow for a cap-and-trade program, the Bush EPA reversed the “appropriate and necessary” finding and instead decided that mercury from power plants can be regulated under a different section of the Clean Air Act. The final rule instead established a NSPS for mercury emissions from new utility sources, and established a requirement that state implementation plans establish emission rate requirements for existing utility sources. In addition to these emission rates, the EPA established a cap-and-trade program in order to reduce the compliance cost of achieving the mercury reduction (Gayer and Hahn, 2005). Much like they did with the NO_x SIP Call and the Clean Air Interstate Rule, the EPA assigned mercury budgets to the states and allowed them the option to fulfill their implementation plan requirements by participating in the national trading program or by devising their own methods in their implementation plans. As with the nitrogen oxides trading rule, states have discretion on how to allocate the allowances to firms, including the option to sell them through an auction.

The mercury cap-and-trade program was designed in part to work parallel with the new sulfur dioxide and nitrogen oxides trading rules for utilities. Because the control technologies to reduce the latter two pollutants yield co-benefit mercury reductions, having the rules follow similar timelines would help firms better coordinate their control planning activities. The mercury trading rule establishes a first phase cap of 38 tons starting in 2010, which according to EPA is achievable strictly through reductions in sulfur dioxide and nitrogen oxides imposed by the other trading rules. Thus, the real costs of the mercury trading rule stem from the lowering of the annual cap to 15 tons in 2018. (Utility mercury emissions were 48 tons in 1999 and thought to be rising without any regulatory intervention). Like the other cap-and-trade

programs, the new rule allows banking but does not allow borrowing of future allowances.⁴

Perhaps the biggest efficiency concern with using a cap-and-trade approach to regulate mercury is that it will result in hot spots, which are localized concentrations of emissions stemming from plants within a small geographic area. Because the rule's cap-and-trade system only restricts the overall emissions level, it is possible that high localized concentrations will result in some areas. Localized concentrations would be a concern because mercury is a known neurotoxin.

Because mercury emissions from utilities present a health risk only through consumption of contaminated fish (NRC, 2000), however, hot spots would only be an issue if they occurred near a water body that contains fish that are later eaten. Because only 34 percent of domestic mercury emissions deposit locally (EPA, 1997), and because much of the fish Americans consume come from sources weakly connected to domestic utility emissions (Gayer and Hahn, 2005), the hot-spot concern is not likely to pose a significant problem. Nonetheless, there remains a great deal of uncertainty about the distribution of mercury deposition, which leaves many citizens concerned about the localized concentrations that might result from a cap-and-trade approach.⁵

The mercury rule offers yet another recent example of how the EPA grafts market-based regulations onto the existing Clean Air Act's framework. This case presented a greater challenge (one which is currently being legally contested), because it entailed regulating mercury under a different section of the statute than was originally advocated by the EPA. There has also been a controversy over the decision to use a cap-and-trade approach (even if such an approach were considered consistent with the Clean Air Act), the appropriate level of the cap, and

⁴The proposed rule allowed for a "safety valve" in which allowances could be bought at USD2,187.50 each from the government, with the pool of future allowances deducted one-for-one for any safety valve allowances purchased. Because the proposed rule did not explicitly prevent indefinite borrowing at the safety valve price, there is a chance that this provision would have amounted to a de facto emissions tax. However, the final rule eliminated this option.

⁵Studies by the (Electric Power Research Institute, 2004, 2005) suggest that neither the cap-and-trade approach nor the MACT approach will lead to hot spots.

the timing of the reductions.⁶ Nonetheless, the cap-and-trade approach that achieves the 70 percent annual reduction by 2018 is expected to save USD15 billion over fifteen years relative to the MACT approach set to obtain a 30 percent annual reduction by 2008 (Gayer and Hahn, 2005).⁷ The cost savings lesson of the Acid Rain Trading Program was a clear motivation for EPA to adopt this approach for mercury regulation.

4.5 Water quality trading

4.5.1 Overview

The effectiveness of cap-and-trade is reduced when there are few traders in the market or pollution is difficult to measure. Both of these problems arise under water quality trading. Nonetheless, there has been a strong and long-standing drive to implement cap-and-trade systems for water quality.

The first major use of an explicit cap-and-trade system in the U.S., for any pollutant, was for pollution of Wisconsin's Fox River. The program covered discharges from sewage treatment plants and pulp and paper mills beginning in 1981. It was not particularly successful, however, due to restrictions on trades and concerns about legal challenges (Hahn, 1989; Hahn and Hester, 1989; EPA, 2001).⁸ Shortly thereafter, water quality trading was also instituted for individual iron and steel plants with multiple outfall pipes, in the form of effluent

⁶ See the interview with Carol Browner in Center for American Progress (2003) for her criticism of the mercury cap and the opinion that the Clean Air Act does not provide authority for a cap-and-trade approach for mercury. See Gayer and Hahn (2005) for a cost-benefit analysis of mercury reduction that suggests a tighter cap would likely not be worth the cost.

⁷ The MACT would result in earlier reductions than the cap-and-trade rule, absent any legal challenges, a difference which suggests that the relative benefits and costs of the two rules will vary by the choice of discount rate. Gayer and Hahn (2005), however, find no change in the relative efficiency of the two rules under discount rates of 3, 5, and 7 percent.

⁸ EPA (2001) also reports that plants developed "compliance alternatives" that were not contemplated when the policy was developed and that potentially contributed to discharges being below the cap. It is not clear whether these alternatives were made possible by the cap-and-trade approach, although the small scale and restrictive nature of the Fox River cap-and-trade suggests that the development of these alternatives was the result of factors outside this regulation.

“bubbles.” These bubbles appear to have been successful in reducing costs (EPA, 2001; Kashmanian et al., 1995) but their applicability to other dischargers was limited. Subsequent trading proposals were initiated by individual states and focused on trading between point and non-point sources, such as for the Dillon Reservoir in Colorado in 1984 and the Tar-Pamlico Basin in 1991. Only recently has a national policy emerged.

The drive for national guidance on water quality trading grew as regulations issued under the Clean Water Act began to take shape. The Clean Water Act of 1972 requires that when watersheds are not meeting their “designated use,” regulators must specify the total maximum daily load (TMDL) that the watershed can handle of each of the offending pollutants. TMDLs are watershed-specific goals that are to be used by regulators to set discharge limits for individual dischargers. Because the determination of a watershed’s TMDLs involves a substantial body of scientific research, and because the analysis needs to be conducted on a watershed-by-watershed basis, it has taken many years for TMDLs to be finalized.

As TMDLs were established, it became clear that meeting them would often require substantial cuts in discharges. Costs were projected to be steep. The idea that these costs might be lowered by allowing trades kindled the drive to develop workable water quality trading systems. In 2001, EPA estimated that such “flexible” approaches to improving water quality could save USD900 million annually compared to the least flexible approach (EPA, 2001).

EPA began seriously considering in 1996 how water quality trading might be allowed under the Clean Water Act, and in the intervening years it has worked on requirements for such programs to operate well. The final rule governing water quality trading policy was issued on January 13, 2003.

Water quality trading as a regulatory approach has been enthusiastically embraced. Breetz et al. (2004) list more than 80 current trading initiatives, in operation or proposed, at all levels of government. Many state-level programs pre-date the EPA rule (Environomics, 1999). Most of the programs that are in operation or have been proposed cover the nutrients phosphorus and nitrogen, but trading schemes have also

been developed for sediment, dissolved oxygen, temperature, stormwater runoff, and heavy metals.

4.5.2 Two examples

Because water quality trading may be less familiar to readers than air trading, we begin this section with two examples.

Trading among point sources. The Long Island Sound nitrogen trading program was devised by the State of Connecticut to control nitrogen discharges from sewage treatment plants.⁹ The system was set up to reduce the costs of meeting TMDL goals for nitrogen flowing into Long Island Sound.

The trading scheme covers 79 publicly owned treatment works (POTW). The set of plants is covered by a general permit that allowed for 16,955 equalized pounds of nitrogen per day in 2003, to be decreased each year, with plant-specific equalization based on the distance of the POTW from the Sound.¹⁰ Each individual POTW was also assigned a share of this total in the form an individual limit which then operates as its initial allocation of nitrogen credits.

The Nitrogen Credit Advisory Board sets the price of nitrogen credits at the beginning of each year based on the observed cost of nitrogen removal projects. At the end of the year, plants that discharged less nitrogen than their individual limit receive a payment for their unused credits from the Board based on the pre-set price. Plants that discharged more nitrogen than their limit allows must pay the Board for the required credits.

There were 40 municipalities that purchased nearly one million pounds of credits in 2003 and 37 municipalities that sold credits. More credits were sold than purchased. The state, through the Nitrogen Credit Advisory Board, purchased the excess credits. Note that the Nitrogen Credit Advisory Board must be prepared to purchase excess credits because the state rather than the market sets the price. So far,

⁹This discussion is based on Connecticut Department of Environmental Protection (2003, 2004, 2005); Kieser and Fang (2005); and personal communication with Gary Johnson, CDEP.

¹⁰All numbers in this paragraph refer to equalized pounds of nitrogen. See next section for discussion of the use of equivalency factors to achieve equalization.

the state has not had to deal with the problem of excess demand for credits at the given price.

Trading between point and non-point sources. The Tualatin River watershed, around Portland Oregon, does not currently meet its TMDLs for dissolved oxygen, phosphorus, bacteria, or temperature.¹¹ The only point source discharger in the watershed is Clean Water Services, a wastewater and surface water management utility that operates four plants plus a storm sewer system. Clean Water Services has applied for a permit that would allow it to trade among plants and also with unregulated non-point sources to meet its TMDL allocations.

To meet its TMDL allocation for temperature, Clean Water Services has claimed that it would need to refrigerate its discharges, which would require a USD35 million capital investment and USD1 million annually in operating costs. A proposed trade, which must be federally approved, would allow Clean Water Services to meet its temperature obligations by: (i) paying farmers and other landowners to plant shade trees along 25 miles of the river; (ii) “persuading” irrigation users to irrigate with warm effluent rather than cold water from a nearby lake, allowing the utility to release less warm water into the river; and (iii) augmenting the river’s flow in the summer with cold water from the lake.

The tree-planting is expected to cost the utility roughly USD6 million. Those farmers would also receive money from other federal conservation programs (thus, the USD6 million is not the full cost of the action; on the other hand, tree-planting also provides non-water-temperature benefits.) There are no cost figures available for the other actions.

To meet its TMDL allocation for dissolved oxygen, the four treatment plants operated by Clean Water Services would be treated as a “bubble,” with a limit falling only on total dissolved oxygen loadings. Thus, the utility would trade pollution implicitly among its own plants.

The proposed trade does not cover Clean Water Services’ discharges of the other pollutants.

¹¹This discussion is based on Environment Reporter (2003).

4.5.3 Water quality trading rules

Water quality trading is made complicated by the large number of water pollutants that are regulated, difficulties in assessing pollution from nonpoint sources, and the small number of traders present in most watersheds. The EPA has issued guidelines that attempt to work around these problems. Much of the discussion around water quality trading presumes that the trades will take place between point and non-point sources (see Section 4.5.5.2), but the EPA's rules apply to both point-point and point-non-point trading.

EPA regulates over 700 forms of water pollution. To be eligible for trading, EPA rules specify that sources be discharging the same type and form of pollutant. In some cases, trading is permitted between two different forms of a pollutant if there is sufficient information to establish *translation ratios* that describe how those pollutants interrelate.¹²

If the pollutant is discharged at different points along a waterway, then EPA also specifies that *equivalence ratios* be established to reflect the distance and hydrology between sources. Equivalence ratios help ensure that each source's ultimate effect on the watershed receives the appropriate weight.

Several trading requirements stem from the way EPA regulates point sources that discharge directly into water bodies. Major point source dischargers are required to have a permit, called the National Permit Discharge and Elimination System (NPDES) permit, which specifies the source's allowable discharges, including the type, form, and timing. Such discharge permits may specify limits on daily, monthly, or yearly loads. Limits on both daily and monthly loads are common for some pollutants. This temporal dimension to pollution control is needed both because of how water pollution affects the ecosystem and because of temporal variability in discharges. Both peak and average loads may affect water quality. See Tietenberg (1985) or Baumol and Oates (1988) for discussion of design of a cap-and-trade system under these circumstances.

¹²This discussion is based on the EPA's Water Quality Trading Assessment Handbook (2004), hereafter called the EPA Handbook.

EPA requires that trades be between sources that have similar discharge timing. A point source facing a monthly limit will be allowed to trade only with a discharger who can demonstrate reductions in all months. This rule may also affect trade between point and non-point sources. Some non-point sources have highly seasonal discharges, so the EPA's "similar discharge timing" requirement would likely rule out some point-non-point trades.

Finally, the NPDES permits are typically valid for five years, after which they must be renewed. EPA recommends that trades between two point sources be approved only when the sources' permit renewal dates are close.

In summary, EPA's rules mean that each trading system must be specially designed based on the waterway and the sources that discharge into it. In theory, all possible trades could be assessed *ex ante*, with appropriate translation and equivalence ratios established ahead of time so that sources would know the basis of any candidate trade. In practice, most trades will have to be individually assessed and approved by the regulators. Only in a few routine cases, such as the Long Island Sound program, can trades be made without case-by-case oversight by regulators.

Because of the specificity of these rules and their informational requirements, substantial bureaucratic effort is involved in setting up a water quality trading system. Regulators will want to know whether such effort is worthwhile.¹³ EPA guidelines state that administrators should determine if "the supply of and demand for pollution reduction credits is reasonably aligned within the watershed," a requirement that will strike economists as odd. An entire section of the EPA Handbook is dedicated to determining the "financial attractiveness" of trading. This focus is intended both to aid regulators in deciding whether to initiate a water quality trading program and, perhaps more importantly, help them spur potential traders to start trading.

¹³ Recall, however, the likelihood that market mechanisms will yield unexpected savings. Early analysis of the iron and steel effluent bubbles suggested that the bubbles would not be particularly valuable, yet subsequent analysis showed them to have generated substantial cost savings. See EPA (2001, p. 100).

4.5.4 Trading between point and non-point sources

Non-point sources are significant contributors to water quality problems, especially high nitrogen and phosphorus concentrations, the pollutants that have been the object of most water quality trading programs. Non-point sources would therefore appear to be important participants in a trading system. The majority of non-point sources, however, are farms, which are not required to hold NPDES permits and whose polluting activities have traditionally gone unregulated. A trading system offers an indirect opportunity to control these discharges, commonly referred to as loadings.

A second reason to include non-point sources is that their control costs are believed to be lower than for point sources (for example, Hall and Howett, 1994; EPA, 2004). Expanding the trading program to non-point sources can also increase the number of potential trading partners. Water quality markets are typically thin because there are rarely a large number of polluters on a given waterway, but including non-point sources could thicken the market and make it more competitive.

Pollution loadings from individual parcels are not measured or monitored, of course; the extreme difficulty of doing so is what leads them to be designated non-point sources. Additional trading rules are therefore needed when non-point sources are involved. Instead of being monitored, the loadings from any particular parcel are *estimated* based on soil type, climate, hydrology, historical land use patterns, previous crop rotations, and how the site is irrigated and fertilized (King and Kuch, 2003).

The sale of a pollution credit by a non-point source therefore stipulates that the landowner will adopt certain (new) land use actions, called Best Management Practices (BMP), which have been estimated to lead to lower amounts of pollutant runoff than current land use practices. That is, the sale of a credit entails a change in land use practices and an accompanying estimate of the associated change in nutrient loadings, rather than an explicit, measured change in loadings. King and Kuch (2003) refer to this as activity-based rather than performance-based criteria.

Uncertainty about the actual non-point loadings has been incorporated into some point-non-point trading rules through equivalence ratios that discount the predicted non-point reductions. The Tar-Pamlico program in North Carolina, for example, uses a 3:1 trading ratio for non-point trades with cropland and a 2:1 trading ratio for non-point trades with livestock farms. The 3:1 ratio means that a one unit increase in discharges from a point-source requires a three unit decrease in modeled loadings from the non-point source. Several economists have explored the optimal trading ratio (Malik et al., 1993; Woodward, 2001).

Non-point sources are not federally regulated, so they start off exclusively as sellers of credits. They have credits to sell if they undertake the land use practices that are approved under the trading scheme. Point-sources, whose discharges *are* subject to regulation, are potential demanders of the credits that non-point sources create. Of course, if a non-point source sells its credits by agreeing to a land use change, it can then be required to act as a buyer of credits if it wants to revert to its former practices.

Inclusion of non-point sources in water quality trading is valuable for two additional reasons beyond the cost savings and the thickening of the market.

(1) Financing of pollution control. The U.S. has traditionally not directly regulated land use at either the State or Federal level.¹⁴ In the Tualatin river case, for example, none of the landowners is required to have an NPDES permit nor are most of their land use decisions subject to EPA approval for their water quality consequences.

Instead, the typical approach throughout the U.S. has been for governments to use economic incentives to encourage environmentally-friendly behavior. In short, the U.S. has traditionally *paid* non-point sources to undertake conservation, improve wildlife habitat, and, in the

¹⁴ Maryland appears to be the first state to regulate non-point loadings from farms, although this law, the Water Quality Improvement Act, is in its infancy and its impact unclear. It requires that landowners file nutrient management plans, but the law's authority either to require the plan to include specific loading-reduction activities or to require the landowner to adhere to the plan is unclear. The law does not allow farms to purchase credits if they want to increase loadings.

current context, adopt BMPs, through Federal programs such as the Conservation Reserve Program, Water Quality Incentive Program, and Wildlife Habitat Incentive Program, generically known as conservation programs. Similar state programs exist in many states. The Federal financing mentioned in the Tualatin case is an example of a conservation program payment.

A water quality cap-and-trade system thus provides a source of additional financing to pay for adoption of Best Management Practices. When point sources buy pollution reduction credits from non-point sources, they are paying non-point sources to control their loadings rather than a State or Federal government doing so through one of the conservation programs.

(2) Entrepreneurship in pollution control. The options for landowners to control their loadings are complex and varied. Many landowners may be reluctant to undertake Best Management Practices even when they are paid to do so. It seems likely that point sources will be more creative and aggressive in looking for reductions than a conservation program administrator might be. This entrepreneurial activity by the point-sources is another way in which water quality trading may enhance non-point source regulation. That private entities will be more aggressive in finding and enrolling cost-effective non-point-source trades than a government program manager is only an hypothesis, however. It has not been tested to our knowledge.

The ability of point sources to recruit non-point sources may be limited or non-existent in practice, however. In North Carolina's Tar-Pamlico program, point-sources do not directly contact the non-point sources. Instead, they make a payment to the Agricultural Cost Share Program, which then contracts with the non-point sources.

4.5.5 Assessment

Water quality trading consists of two very different kinds of markets: markets consisting mainly of point sources whose discharges can be monitored and who trade on a frequent basis; and markets consisting of a few point-sources who trade with non-point sources, with trades typically occurring on an infrequent basis. We assess these separately.

4.5.5.1 Point-point trading

Point-point trading would appear to lend itself to successful cap-and-trade. There are few active point-point trading systems in the U.S., however. The two programs that have received the most attention are the Long Island Sound and Tar-Pamlico programs.

The Long Island Sound program covers nitrogen discharges from 79 point sources in a fairly smoothly operating market. It uses a Nitrogen Credit Advisory Board, however, rather than the market, to set credit prices. The Board sets the price of a credit based on the observed capital and operating costs of nitrogen removal. The stated rationales of this pricing-setting rule are to minimize transaction costs and exert “overall market control” (Kieser and Fang, 2005). Neither of these rationales are likely to strike economists as compelling.

The Tar-Pamlico program, which covers both nitrogen and phosphorus, allows point-point trades at privately negotiated prices but it also offers a safety-valve price. In the event that aggregate point-source discharges exceed the cap, load reductions are purchased from non-point sources at the safety-valve price. Payments are made to the Agricultural Cost Share Program, which uses those payments to finance farmers’ adoption of best management practices. Hoag and Hughes-Popp (1997) note that the safety-valve price is set using the average cost of BMPs rather than marginal cost, which introduces a potential distortion. This distortion applies only to the pricing of non-point source credits; point-source reductions would still be cost-effective.

Nitrogen and phosphorus from point sources are particularly suitable for trading. Discharges can be monitored and the pollutants tend to have downstream rather than localized effects, so that peak-load problems are less important. Spatial considerations are rather easily dealt with in the form of equivalence ratios.

Point-point trading has also been facilitated by the performance-standard approach of existing water pollution regulation, which allows a substantial degree of within-plant flexibility and is relatively easily extended to across-plant trading. Only a small change in the permitting process is required: a shift from the issuing of individual permits that

limit discharges at each point source to the issuing of “general permits” that limit total discharges over a set of point sources.

Despite these favorable conditions, point-point trading schemes are rare. In many watersheds the number of point-source dischargers is too small for trading to be worthwhile. Most pollutants other than nitrogen and phosphorus are not easily monitored and are likely to exhibit local or peak-loading problems.

Because water quality trading requires local initiative and watershed-specific design, potential proponents may be unfamiliar with trading’s benefits. For both the Long Island Sound and Tar-Pamlico programs, the presence of a safety-valve or state facilitation of market-clearing seems to have been key to market stability and therefore acceptance of the program by sources.

Finally, somewhat bewildering for economists is the view of cap-and-trade by the actual participants: The Long Island Sound programs attributes its success to public funding of water treatment investments (CDEP, 2004); not, apparently, to sources’ ability to trade.

4.5.5.2 Point-non-point trading

Considerably more attention, by policymakers, interest groups, and researchers, has been focused on trading between point and non-point sources. The thorough and authoritative discussion of nutrient trading by King and Kuch (2003) focuses almost exclusively on point-non-point trading. Kieser and Fang (2005) provide the most up-to-date review of trading but focus primarily on point-non-point trades. Woodward and Kaiser (2002) review the institutions that facilitate trading, but again focus on point-non-point trades. This excessive attention may come at the expense of attention to improving the performance of the more suitable point-point trading.

The inclusion of non-point sources in a cap-and-trade is motivated by potential cost savings and by prospects for a thicker, better-functioning market. We believe that non-point source trading achieves some cost-savings but essentially no improvement in market function.

In most cases, the trades must be individually approved. This requirement stems from the nature of non-point source pollution control and is not a regulatory artifact. Also, point-non-point trades are infrequent and may become even more infrequent over time as fewer and fewer willing landowners or eligible parcels remain in the trading set. In selling a credit, a landowner agrees to undertake land use practices that may last many years, such as planting trees. Therefore, many trades are essentially permanent transactions.

Even the cost savings must be treated with some skepticism because of the unknown effectiveness of the prescribed land use practices and the possibility that some of the non-point control actions would have been taken in the absence of trading. Note that the transactions costs are also substantial.

A remaining question is whether it is possible to shore up these weaknesses and improve the efficiency of non-point trading. Hanson and McConnell (2005) suggest focusing on a small number of readily observable, annual BMPs, such as the planting of winter cover crops, which take up excess nutrients and reduce winter erosion. The use of such crops to reduce non-point loadings is a single-year action that can be easily verified. Therefore, trading can proceed with greater regulatory, less oversight and less regulatory uncertainty, features which should lead to much thicker markets. The sacrifice is that these may not be the lowest-cost actions.

The cost-savings available from including non-point sources in water quality trading may be further curtailed by political opposition. Agricultural interests and other public interest groups frequently want to preserve land in agriculture, a goal which leads them to oppose the allowing of trades that result in significant quantities of land being taken out of production. This restriction diminishes the cost savings of non-point trading.

Non-point trading is especially valuable because non-point sources are believed to provide lower-cost pollution control and trading is the only way (at present) to bring in these sources who would otherwise be unregulated. The long-run prospects for this approach are unknown. Farming practices will change, as will our understanding of the science behind non-point source pollution and the connection between land

use practices and non-point loadings. Changes to the trading rules will almost certainly be needed at some time.

The implications of this evolution in the understanding of non-point pollution have not been much explored. First, the reductions that can be achieved by non-point controls may be found to be higher or lower than current estimates, thus changing the relative marginal costs and the desirable mix of point and non-point reductions. Second, the specific practices that are found to lead to reductions in non-point loadings may change, thus changing the nature of point-non-point trading. For example, certain types of farms in particular locations may turn out to be more valuable sources of non-point reductions than currently believed. Likewise, the kinds of farming practices that are believed to yield reductions may also change.

In other words, the nature of water quality trading and the projected gains from trade may change substantially as new scientific information becomes available about the relationship between land use and non-point erosion and nutrient loadings.

A related problem is that if ambient water quality does not decrease by the credited amount, there is no penalty to the non-point sources (who may have profited from sales of pollution credits) under current rules. Indeed, it appears that any burden would eventually fall back on the *point sources*, which would face stiffer regulation as a way of meeting the TMDL. Note that point-non-point trading ratios attempt to minimize this possibility but they do not hedge the risk to point-sources. On the other hand, if TMDLs are not met, states may eventually be compelled to directly regulate non-point sources.

4.6 Environmental taxes

Although environmental taxes share most of the advantages of cap-and-trade, the role they play in shaping environmental quality appears to be much smaller. EPA (2001) argues that such taxes and fees “tend to be set at rates too low to have a significant impact on pollution” EPA (2001, p. iv). This situation raises the question of how those taxes, as currently designed, would perform if they were higher. In every instance we find, environmental charges are levied on top of other regulations

and are not used as the primary mechanism to control pollution. Under such circumstances, many of the advantages of environmental taxes will not be experienced.

4.6.1 Types of taxes

The number of environmental-type taxes in the U.S. is large (see EPA, 2001). This section covers five categories that are particularly prominent or provide useful economic lessons. The lessons are covered in Section 4.6.2.

4.6.1.1 “Environmental taxes”

The U.S. tax code defines four taxes as “Environmental Taxes”:

- (1) Excise tax on crude oil, at a rate of 14.7 cents per barrel.
- (2) Excise taxes on “certain chemicals.” The tax on ammonia is USD2.64 per ton. The tax on acetylene is USD4.87 per ton. Currently, 42 chemicals are taxed.
- (3) Excise taxes on “certain imported substances.” This section covers the importation of chemicals described above.
- (4) Excise taxes on ozone-depleting chemicals. A base tax of USD9.85 (2005) is multiplied by the chemical’s ozone-depletion factor.

These taxes are quite small. A barrel of oil yields 42 gallons of gasoline, so the crude oil tax works out to roughly one-third of a cent per gallon of gasoline. The gas tax, which has almost identical effects (but is not labeled an environmental tax), is much higher than this. For ammonia, quoted prices are as low as USD390 per ton, so the ammonia tax is about 0.6 percent of the price. For acetylene, quoted prices are as low as USD360 per ton, so the acetylene tax is about 1.4 percent of the price.

The feature that leads these rather minor taxes to be explicitly labeled *environmental*,¹⁵ while others with greater impact are not, is

¹⁵A portion of the corporate income tax is labeled an environmental tax, but the tax has nothing to do with the environment.

the assignment of the tax revenue. The oil and chemical taxes are assigned to trust funds (e.g., Oil Spill Liability Trust Fund, Hazardous Substance Superfund) with an explicit environmental damage amelioration purpose. Gasoline and coal severance taxes implicitly target the same externality as the crude oil tax and are larger in magnitude, but their revenue is not assigned to an environmental fund. They therefore are not labeled as environmental taxes by statute. The one exception is the ozone-depleting chemical (ODC) tax, which is paid into general revenue. The ODC tax is discussed in Section 4.13.

Berthold (1994) describes the gas-guzzler tax (see next section) and ozone depleting chemicals tax as the only taxes that at all resemble the textbook Pigouvian tax.

4.6.1.2 Federal taxes related to automobiles

Several Federal taxes fall directly or indirectly on externalities generated by automobiles but are not labeled environmental taxes:

- Gas-guzzler tax. This is an excise tax on new car sales based on the model's fuel economy. The tax increases with decreases in fuel economy. A model with greater than 22.5 miles per gallon (mpg) faces no tax. A model with at least 21.5 mpg but less than 22.5 mpg faces a tax of USD1,000. Vehicles with less than 12.5 mpg face the largest tax, USD7,700. Light trucks such as minivans and sport-utility vehicles are excluded from the tax.
- Tire tax. This is an excise tax on new tire sales based on the tire's weight.
- Vehicle tax. This is similar to the gas-guzzler tax but it (i) applies each year and (ii) is based on vehicle weight, not mileage. The tax is 0 for vehicles below 55,000 pounds, USD100 per year plus USD22 for each 1,000 pounds above 55,000 pounds, with a cap of USD550 per year for vehicles over 75,000 pounds.

The gas-guzzler tax is similar to fuel economy standards (discussed in the next section) in that it applies only to new purchases, exempts

light trucks, and is tied to mileage. It raised roughly USD71 million in 2000. Only a few studies have examined its economic effects (see citations in Greene et al., 2005). Greene et al. (2005) argue in favor of a close and potentially superior alternative, called a feebate, in which new vehicles below a pivotal mileage standard pay a tax but new vehicles above the pivot receive a rebate, with the tax or rebate amounts increasing in the distance from the pivot standard.

The tire tax raised USD420 million in 2000. The tire tax seems a poor conduit for environmental policy because its link to environmental externalities is weak. The tax may be expected to reduce tire demand, but the link between tire demand and miles driven or gasoline consumed, *ceteris paribus*, is unknown. If the concern is tire disposal, a tire tax is superior to a levy on tire disposal, because of its immediacy and the problem of illegal disposal.

The vehicle tax differs from many of the other taxes or regulations applied to transportation in its being applied annually rather than only to new vehicles. Partly because of this feature, it raises considerably more revenue than the other taxes in this section, USD893 million in 2000. The effect of the tax on gasoline consumption is unknown.

4.6.1.3 State hazardous waste taxes

At least 30 states impose taxes on hazardous wastes. Most of these taxes are levied at the point of final disposition, which may be a landfill, incinerator, underground injection site, or recycling center. California also levies a tax when the waste is generated. Taxes may vary by medium and the toxicity of the waste. Sigman (1996) remarks that these taxes are far from Pigouvian, since the rates are based on revenue needs rather than environmental costs.

Hazardous waste taxes have been subject to more analysis than other environmental taxes. Nonetheless, the complications are vast and the lessons unclear. Taxes affect the method of disposal, the disposal location (including transportation of varying distances), and toxicity. The relationship between these elements is complex. Each of these items also has its own externality. Therefore, the overall efficacy and efficiency of the taxes is difficult to assess (Levinson, 1999; Sigman, 1996).

4.6.1.4 Direct discharge fees for effluent

Hahn (1989) lists direct discharge fees on wastewater as the (sole) application of environmental charges in the U.S. In 1993, 39 states assessed fees to point-source dischargers, mostly municipal wastewater treatment plants, which held (non-tradable) NPDES permits, described in Section 4.5. Ten of these states assessed those fees based on discharge volume and pollutant concentration, while 18 states assessed fees based on discharge volume alone (EPA, 2001). The Clean Water Act allows states to take over enforcement of Clean Water Act regulations, including issuance of NPDES permits.

EPA (2001) notes that the incentive effects of water effluent fees have not been studied in a comprehensive way, although the effects are believed to be small (Boland, 1986; EPA, 2001). A substantial empirical literature on water pollution discharges exists (see literature reviewed in Bandyopadhyay and Horowitz, 2006; Earnhart, 2004; and McClelland and Horowitz, 1999), but none of these articles has found fit to examine the role of effluent taxes, presumably because they are simply too small to matter.

4.6.1.5 Energy taxes

Gasoline taxes are levied by both the Federal government and the states. The Federal tax is an excise tax currently at 18.4 cents per gallon. This is a major excise tax that generates more than USD20 billion per year. The history of the tax is shown in Table 4.1. Figure 4.1 depicts the tax in nominal and real terms and shows the wide swings in the real price over time. The Federal Highway Administration (2006) provides an interesting history of the gas tax that describes how it was introduced by President Hoover to head off a Federal budget deficit during the Great Depression.

The state taxes take various forms. Some states levy an excise tax, whereas others levy an *ad valorem* tax. New Jersey levies both. The (weighted) average state tax was 19.1 cents per gallon as of December 2004, roughly equal to the Federal tax, but this average conceals a large range across the states, from 7.5 cents in Georgia to 32.0 cents

Table 4.1 Federal Gasoline Tax (Cents per Gallon).

Date	Nominal tax	Revenue use (Percent of revenue)			
		General revenues	Highway account	Mass transit	Other
June 21, 1932	1.0	100.0	0.0	0.0	0.0
June 17, 1933	1.5	100.0	0.0	0.0	0.0
January 1, 1934	1.0	100.0	0.0	0.0	0.0
July 1, 1940	1.5	100.0	0.0	0.0	0.0
November 1, 1951	2.0	100.0	0.0	0.0	0.0
July 1, 1956	3.0	0.0	100.0	0.0	0.0
October 1, 1959	4.0	0.0	100.0	0.0	0.0
April 1, 1983	9.0	0.0	88.9	11.1	0.0
January 1, 1987	9.1	0.0	87.9	11.0	1.1
September 1, 1990	9.0	0.0	88.9	11.1	0.0
December 1, 1990	14.1	17.7	70.9	10.6	0.7
October 1, 1993	18.4	37.0	54.3	8.2	0.5
October 1, 1995	18.4	23.4	65.2	10.9	0.5
January 1, 1996	18.3	23.5	65.6	10.9	0.0
October 1, 1997	18.4	0.0	83.9	15.5	0.5

Source: Buechner (2005).

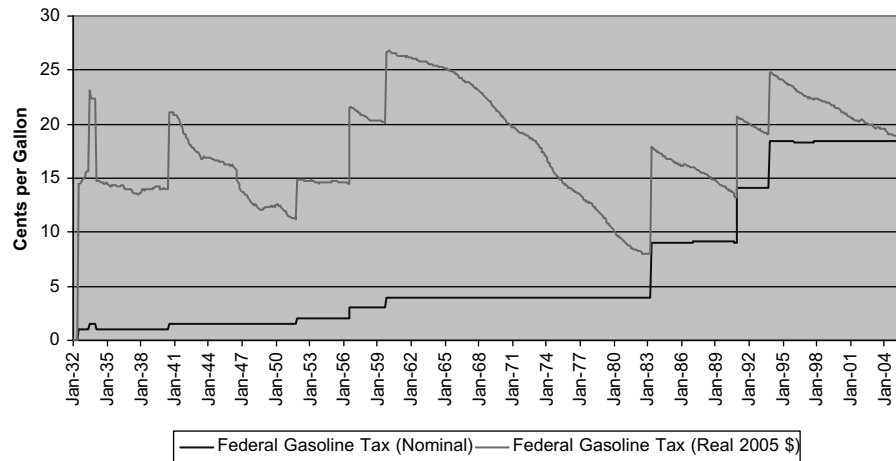


Fig. 4.1 Federal gasoline tax (1932–2004).
Source: Congressional Research Service.

in Connecticut. Some states also allow local communities to levy additional gas taxes.

Different motor fuels are taxed differently. Federal motor fuel taxes are levied on gasoline (18.4 cents per gallon), diesel (24.4 cents per gal-

lon), liquefied petroleum gas (13.6 cents per gallon), compressed natural gas (48.54 cents per thousand cubic feet), and gasohol (13.2 cents per gallon). This distinction also applies to the state taxes. State taxes also contain various exemptions and special provisions; for example, liquefied petroleum gas users in California may opt to pay an annual fee in lieu of the volume tax.

Oil, natural gas, and coal are subject to state severance and *ad valorem* taxes, and coal is subject to a Federal severance tax. A severance tax applies only on extraction. Therefore the tax will have strong effects on energy supply only when a large proportion of the fuel is produced domestically, which is true for natural gas (about 95 percent of U.S. consumption is extracted domestically) and coal (more than 98 percent of U.S. consumption is mined domestically), but not crude oil (about 30 percent of U.S. consumption from domestic sources). There is no Federal severance tax on oil or natural gas.

The Federal government does levy multi-faceted severance taxes on coal as follows: (i) USD1.10 per ton for non-lignite coal from underground mines; (ii) USD0.55 per ton for non-lignite coal from surface mines, and (iii) no severance tax for lignite coal. The 2005 price of coal on the New York Mercantile Exchange was around USD58 per ton, so the taxes are in the range of 1 to 2 percent of the price. The coal severance tax generates just USD500 million per year.

This paper's analysis has focused on Federal environmental policy, but in the case of natural gas, most of which is extracted from a small number of states and is subject to state severance taxes, state taxes may be important. Over 50 percent of U.S. production comes from Texas and Louisiana. Texas levies a 7.5 percent statewide severance tax and its counties also levy *ad valorem* taxes. Louisiana levies an excise tax ranging from 1.3 to 12.2 cents per million cubic feet.

Energy taxes are of particular interest now because they can be easily converted into carbon taxes, using the carbon contents of the various fuels (see Barrett, 1991; Poterba, 1993).

4.6.2 Assessment: Taxes as environmental policy

An extremely large literature exists on the theory of environmental taxes, as Section 3 demonstrates. This literature is exceptionally large

given the small *regulatory* role that taxes play. Indeed, it is essentially impossible to point to an environmental tax with any regulatory role at all. The strongest example is Renewable Portfolio Standards, which appear in fact in the guise of cap-and-trade! (see Section 4.8.) There is no instance in the U.S. in which taxes have been used as the sole instrument to control pollution.

Because tax rates are low and tend to be issued at the state level, real-world taxes other than gasoline taxes have not been subject to much analysis. A related point is that the states have played a relatively larger role in the use of environmental taxes compared to other policies. Generalizations about the performance of environmental taxes are difficult to make given this limited analysis and limited U.S. experience. In particular, economists have written little about how the taxes interact with the other regulations that have been used to govern the various activities being targeted. The exception is gasoline taxes, for which a very large literature exists.

Just a few lessons can be drawn concerning the interaction of taxes with existing regulations. Ozone-depleting chemical taxes were applied on top of a cap-and-trade system for the same chemicals. Because the expectation was that the cap would be binding, the tax's intended purpose was to capture the windfall that the cap gave to existing producers of ODCs (Hoerner, 1997). Alternative chemicals and processes were developed relatively quickly, however, and the cap was not binding for the first four years after its implementation, which gave the tax a desirable and unanticipated role in further reducing chemical use.

Water pollution has historically been governed by performance standards that assign limits to pounds of effluent per month or maximum effluent concentrations from individual sources, for example. Taxes levied on top of these standards would appear to improve efficiency because most sources operate below the performance standard even when the variability of discharges is accounted for (Bandyopadhyay and Horowitz, 2006). Taxes may have the potential to perform better than cap-and-trade for point-source water pollution because they avoid the problems that arise under thin markets.

Energy taxes raise the per-unit price of fuels or electricity but are also part of a web of regulations and tax incentives targeted as energy

use. Regulations primarily take the form of energy efficiency standards for buildings and appliances and fuel economy standards for cars. Such efficiency standards are inefficient because the marginal cost of energy use is not equalized across new and existing sources. Parry et al. (2004) examines the interaction of gas taxes and CAFE standards.

Hazardous waste taxes are imposed in conjunction with federal treatment standards issued under the Resource Conservation and Recovery Act, among other standards. In this situation, taxes create complicated and possibly perverse incentives. If treatment standards render the waste benign, then taxes are inefficient because there would be no need to cut back on the waste. If treatment standards do not render the waste benign or, more generally, do not equalize the toxicity of the wastes, then taxes and treatment standards create incentives for cross-chemical and cross-media substitution. In such cases, taxes may not add to the efficiency of a hazardous waste disposal policy.

In summary, taxes, to the extent that they are used, are in every instance levied in conjunction with other regulations. General economic principles may guide economists in thinking about the interaction of taxes and other regulations but there is little empirical analysis or detailed policy analysis to supplement this intuition.

A further lesson applies to the efficiency property of an environmental tax, rather than its more fundamental cost-effectiveness property. Efficiency requires that the tax be equal to the marginal value of the negative externality. From a practical point of view, since damage functions are difficult to estimate, one might expect that the tax would at least be qualitatively related to the value of the externality: a high value externality should be associated with a higher tax than a low value externality. A third criterion is that a change in the perceived value of an externality should be associated with a change in the tax in the same direction.

The gas tax clearly fails even the last and weakest of these criteria. The debates surrounding the increases in the federal gas tax in 1990 and 1993 (see Figure 4.1) focused on federal deficit reduction and the division of increased tax revenues between the General Fund and the Highway Tax Fund (CRS, 2000), not any increased concern over externalities from gasoline. Given the salience of the climate change problem

in the late 1980s and early 1990s, as evidenced by the administration's signing of the U.N. Framework Convention on Climate Change in 1992, it is telling that climate change was not a stated justification for the 1990 and 1993 gas tax increases. This absence suggests that taxes were not seen as a potential tool for environmental policy.

4.7 CAFE standards

4.7.1 Overview

In 1975, Congress passed the Energy Policy and Conservation Act, which established Corporate Average Fuel Economy (CAFE) standards. CAFE standards set miles per gallon limits for new passenger cars sold in the U.S. The standards are to be met on a fleet-wide average. The initial standard was 18 miles per gallon for model year 1978, and the statute established a partial schedule for attaining a goal of 27.5 miles per gallon by 1985. Because CAFE applies on a fleet-wide basis, this rule implicitly sets up a within-company cap-and-trade. Legislators could have extended this approach to cross-company trading, that is, a true cap-and-trade system, but did not, despite the fact that there were several other Federal forays into cap-and-trade around this time.

Instead, Congress opted to move in the opposite direction, by creating a separate CAFE standard for light trucks, a category which at the time almost entirely consisted of pickups but has since grown to include a large number of minivans and sport-utility vehicles. Rather than establish the light truck CAFE standard by statute, Congress directed the National Highway Traffic Safety Administration (NHTSA) to determine the standards. The CAFE standard for light trucks is currently 20.7 miles per gallon, which is a less restrictive standard than for cars, thus creating an implicit subsidy for light trucks.¹⁶

¹⁶ A recent Notice of Proposed Rulemaking issued by NHTSA outlines a new system of CAFE for light trucks in which different standards apply to light trucks with different "footprints" (track width multiplied by wheelbase). The smallest category would face standards similar to passenger cars, thus eliminating the incentive to build and categorize vehicles as "light trucks." The proposed standard is more relaxed for each larger light truck category (Federal Register 05-17005, Vol. 70, No. 167).

4.7.2 CAFE's market-based features

CAFE standards offer substantial within-firm flexibility to producers because they are fleet-wide standards. A manufacturer desiring to meet the standard has the option of improving the fuel economies of its vehicles or lowering the prices of vehicles that are above the standard. If the manufacturer can sell enough of the latter vehicles to bring the weighted average to the CAFE standard, then it is in compliance. In this way, CAFE sets up a cap-and-trade system that operates within each manufacturer. However, there is no trading across fleets and thus there are no credit "prices."

An additional market-like feature is that manufacturers earn credits if they produce below the CAFE standard in any given year and these credits can then be banked. However, the credits cannot be sold to other manufacturers and they expire within three years. Credits earned for a passenger car fleet cannot be applied by the manufacturer to its light truck fleet, and vice versa.

Another market-based feature of the CAFE program is its penalty of USD5.50 per new vehicle sold for each 0.1 mile per gallon that a fleet falls short of the standard. This essentially establishes a safety-valve for manufacturers, since they can opt to pay the fine rather than meet the fleet-wide cap. A few European manufacturers of luxury cars have opted to pay the fine rather than meet the CAFE standard. Economists typically treat penalties and fees as equivalent to taxes, but some authors have argued that the label has real-world consequences (Berthold, 1994; Goodin, 1994). The CAFE example appears to be a case where the economists' interpretation is right.

Although the CAFE system contains multiple market-based features, it falls short of a true cap-and-trade program because it does not allow trading across manufacturers. Allowing trading across fleets would reduce the cost of meeting an overall mileage goal because it would allow manufacturers with high marginal costs of high-mileage vehicles (due, for example, to lack of expertise in fuel-enhancing design or to a marketing reputation for particular types of vehicles) to purchase credits from manufacturers with low marginal costs of high-mileage vehicles. A full market-based approach would also include

indefinite banking, which could be especially helpful in lowering the cost of any policy that prescribed sharp increases in the mile-per-gallon standard over time. One further complication to consider for an industry-wide CAFE cap-and-trade is that the automobile market is not perfectly competitive.

4.7.3 CAFE's weaknesses

CAFE has a much greater failing, however, in that it does not cap an identifiable externality. The original goal of CAFE was to reduce U.S. gasoline consumption. Yet, the problem with the CAFE standards is that a cap on miles-per-gallon is not the same thing as a cap on gallons consumed. Thus, the consequences of CAFE must be evaluated not only in terms of the overall costs of meeting the CAFE standard but also in terms of the effect on nationwide gasoline consumption, which is the product of average miles per gallon of both new and existing cars and total miles driven. These consequences are not easy to assess because of (i) the effects of CAFE on the automobile market and therefore on the number of new cars sold and (ii) the so-called "rebound effect" in which a higher mile-per-gallon standard lowers the cost of driving a new car and thus leads to more miles driven (see Jones, 1993; and Greene et al., 1999). In the latter case, although the percent reduction in miles-per-gallon is generally believed to outweigh the percent increase in miles driven, the ultimate effects on gasoline consumption are difficult to predict.

This misplaced-cap feature also makes it hard to predict the effect of any particular mileage standard on aggregate gasoline consumption. When the Clear Skies Act proposes a cap of 4.5 million tons of sulfur dioxide, legislators and the public have a firm and consistent idea about the amount of sulfur dioxide that will be emitted by power plants. When CAFE rules specify a standard of 27.5 miles per gallon, citizens and legislators do not necessarily have consistent ideas about the effect on aggregate gasoline consumption. The number of new cars purchased and the miles that they will be driven play large, accompanying roles in determining gasoline consumption. A pure cap-and-trade for imported

gasoline (if foreign oil dependence were the concern) or for carbon would have the added benefit of being transparent.

One tempting “fix” might be a cap-and-trade for predicted lifetime gasoline consumption of a new car fleet. Such a system would, however, retain many of CAFE’s failings or introduce new ones. This cap-and-trade system would still not give consumers the incentive to consume gasoline efficiently. The rules for predicting lifetime gasoline consumption would require Federal rule-making (rather than simply legislation) and therefore be subject to lawsuits and likely delays. Also, the cap could fail to be binding if consumers purchased a low number of new cars, but this would be an *undesirable* outcome if the consumers were instead holding on to their old, lower-mileage cars.

4.7.4 Safety implications of CAFE

CAFE also fails to address other externalities of driving associated with traffic congestion, non-carbon emissions, and the safety risks of opposing drivers. This section examines the external safety implications of CAFE.

There are three options available to manufacturers to meet the CAFE standards. They can 1) install technologies in their vehicles that improve fuel economy, 2) reduce the weight of their vehicles, or 3) reduce the price of their more fuel efficient (i.e., lighter) vehicles to offset the under-compliance of their other vehicles. There is some evidence to support that the second two options have indeed occurred, which has led to a downsizing of passenger cars (Crandall and Graham, 1989).

The concern of this downsizing is that it could lead to more traffic fatalities, since there is evidence to suggest that two light-weight vehicles in a crash will lead to more expected fatalities than two heavier vehicles in a crash (Evans, 1984). However, there are a few qualifications to consider.

First, from an efficiency perspective, the pertinent regulatory issue is not the total fatalities resulting from the mix of vehicle weights. The risk of a given vehicle is internalized into the consumption decision.

However, the risk of an opposing vehicle does pose an external cost – one which can then justify appropriate government regulations.

Second, the early fatality estimates by vehicle weight were obtained by analyzing cars, not light trucks (which were a much smaller proportion of vehicles sold in the early 1980s). The results may not be applicable because the number of fatalities given a crash is also a function of the weight disparity of the vehicles, which have likely changed as light trucks have become more popular (NHTSA, 1997).

Finally, the fatality estimates are derived conditional on a crash occurring. They do not consider that light trucks (which face a more relaxed CAFE standard) are more likely to crash in the first place (Gayer, 2004). So even though lighter cars stemming from CAFE might lead to more fatalities, the implicit CAFE incentive towards light trucks (which are more crash prone) might lead to more fatalities.

In Section 4.7.3, we argued that CAFE failed to address the main externality it was targeted at, namely gasoline consumption. A cap on total gasoline consumed would better target the externalities associated of fuel consumption, but would still fall short of targeting the externalities associated with traffic congestion, non-carbon emissions, and the safety risks of opposing drivers.

4.8 Renewable portfolio standards

4.8.1 Overview

Renewable portfolio standards (RPS) are a requirement that a specified percentage of electricity sold at the retail level be generated by renewable sources. In most versions of the regulation, generators of renewable electricity are awarded renewable energy credits which can then be traded among electricity providers to meet the RPS. Thus, RPS is a form of cap-and-trade, used to encourage renewable energy.

As of October 2005, 21 states and the District of Columbia had established RPS requirements. At the Federal level, RPS was passed by the Senate in various versions of the bill that was to become the 2005 Energy Bill, but it was never part of the House versions and was not included in the final bill. RPS regulations are also used in Japan and in various European countries.

Under the proposed Senate version, 2.5 percent of the electricity produced in the U.S. in 2008 would have to be from renewable sources.¹⁷ RPS percentages for the state programs range from 4 percent in Massachusetts (by 2009) to 20 percent in California and Nevada (by 2017 and 2015). A few of the states (Iowa, Minnesota, Texas) have a requirement for the quantity of renewably-generated electricity, rather than a percentage. Maine's RPS is 30 percent, but it allows cogeneration to qualify, a source that would be economically viable even without an RPS requirement.

Most of these programs also contain a safety-valve, sometimes called a cost cap or, in the U.K., a "bug-out price." Under the proposed Federal program, the allowance price for renewable energy credits was capped at 1.5 cents per kilowatt-hour, not indexed for inflation. Plants could buy a limitless number of credits from the Federal government at 1.5 cents. These credits would not need to be paid back; that is, they would not be subtracted from future caps. The Massachusetts safety-valve is 5 cents per kWh, and automatically adjusts for inflation. The Texas safety-valve's purpose is particularly transparent: It is the lower of 5 cents per kWh or 200 percent of the previous year's average credit price. Neither the proposed Federal legislation nor any of the state programs allows banking or borrowing of credits.

The state programs also vary in terms of more fundamental trading rules and arrangements. Although RPS appears to translate to a straightforward cap-and-trade program, few of the state programs take advantage of all of the cap-and-trade features. There are three major sorts of state RPS programs: (1) A statewide cap-and-trade program governing retail sellers of electricity, complete with tradable credits. This was also the structure of the proposed Federal program. (2) A statewide electricity tax, sometimes called a Systems Benefit Charge. Revenues are channeled into a fund that then purchases renewable electricity from sources that have applied to be included and been certified. (3) A company-level renewable requirement. A company-level

¹⁷RPS is analogous to state and local laws that require a certain percentage of recycled material in paper, although none of these laws allows this requirement to be met *on average*, to our knowledge.

requirement is analogous to the current CAFE standard, which is also not tradable across companies.

States also differ in the set of sources that are eligible to meet the RPS. Wind appears to be the main beneficiary of the state RPS requirements. Biomass, solar, geothermal, ocean, landfill gas, fuel cells, and small hydro are also valid sources for various states. States may also not treat all renewables as equally valid. Arizona devotes roughly 50 percent of its renewables fund to purchasing solar energy, and Nevada requires that 5 percentage points of its eventual 15 percent RPS be met through solar. Other states such as Connecticut and New Jersey define two classes of renewables and have separate RPS for each.

4.8.2 Discussion

4.8.2.1 Externalities addressed by RPS

Proponents of RPS advance different ideas about the externality problem that RPS is meant to solve. Air quality is the problem most often mentioned and for this reason some states do not allow some forms of so-called “dirty biomass” to qualify. The air quality justification is weakened by the fact that RPS treats all nonrenewables equally in assessing the electricity base, regardless of their air pollution emissions.

Climate change is another commonly mentioned justification. This is problematic for two reasons. First, as with air quality, RPS treats all nonrenewables equally regardless of their carbon content. A source that produces electricity from natural gas is required to generate the same amount of renewable electricity or buy credits as a source that produces from coal.

Second, not all renewables are carbon-free. Biomass (plant material burned to generate steam for electricity) clearly reduces carbon emissions, relative to no-RPS, only if the biomass source does not rely on extraction of a stock of biomass. If a stock of biomass is being extracted then the use of biomass as fuel does not necessarily decrease carbon emissions. Therefore, biomass burning does not have a uniform effect on carbon emissions. RPS rules have not yet attempted to address this issue. The net change to the carbon cycle is a complicated

determination since it is difficult to know what the carbon stock of any given land parcel would be if it were not being used to supply biomass.

Like CAFE, renewables are also mentioned as a way to reduce dependence on foreign energy. Renewable energy is also mentioned as a way to diversify energy sources and protect against oil price shocks (Wiser et al., 2005), but this is valid only if electricity suppliers are not correctly diversifying their energy portfolios. A final common argument is that RPS is useful in fostering the growth of renewable technologies and the recognition of their potential. This argument is only valid, however, in the context of one of the other justifications for renewable fuels.

The externalities that RPS is meant to address should dictate the eligibility of renewable sources. Cogeneration, biomass, or other potential sources are useful for address some externalities but not others. If these externalities were spelled out, a remedy would be a trading ratio for tradable credits from different sources based on the weighted sum of the externalities that each source produces.

4.8.2.2 Price volatility

An early concern with RPS was the price volatility of renewable energy credits. The RPS is believed to make the demand for renewable energy (by electricity retailers) highly inelastic. The newness of some forms of renewable energy and the low capacity of renewable energy producers is posited to make the supply of renewable energy similarly inelastic. The result is predicted price volatility. Such volatility would be costly, raising the cost of capital to electricity providers. Banking and borrowing of credits would alleviate some of this predicted volatility, but RPS rules have so far not permitted banking or borrowing. Given that renewable sources are meant to diversify energy sources, price volatility seems counter-productive.

Such concerns appear to be overblown. Like other cap-and-trade systems, regulated sources may quickly learn to adapt and overcome the problems, particularly in finding alternatives that lead demand and supply to be less inelastic. The safety-valve feature should also be effective in reducing price volatility.

4.8.2.3 Assessment

Like water quality trading, RPS has received a great deal of interest from the states within the last few years. On the whole, renewable energy appears to lend itself well to the RPS approach. Despite the policy's widespread adoption, however, "there does not yet exist a renewable energy credit market of any size" (Chupka, 2003, p. 48). Complex rules, particularly over enforcement and eligibility, have led to delays in the implementation of trading (see Wisser et al., 2005 for discussion of problems in California's RPS). Therefore, an assessment of its performance is not yet possible.

The failure to define a specific externality and to tailor the policy to that externality hinders the broader success of the RPS approach. For externalities such as climate change, RPS falls short of being cost-effective because it does not differentiate between the carbon content of the nonrenewable fuels that are being displaced. A similar argument holds for air quality.

It is important to recognize that RPS is better viewed as a tax (on nonrenewables) than as a form of cap-and-trade. Each kilowatt-hour of nonrenewables must be accompanied by the required percent of kilowatt-hour of renewables, which are more expensive (if the RPS is binding); the implicit tax is the difference in these costs. RPS is not an actual cap-and-trade because nothing is capped; because of a firm's freedom to increase electricity production, the quantity of nonrenewable electricity can actually increase under an RPS.

Because RPS effectively acts as a tax and because many different programs are in effect, RPS may give researchers their best chance to judge the performance of environmental taxes.

4.9 No net loss of wetlands

4.9.1 Wetlands protection under the Clean Water Act

In 1990, the Bush Administration issued a "no net loss of wetlands" rule to govern a type of water quality permit being issued under the Clean Water Act. The term naturally suggests a cap-and-trade approach to wetlands management, although the full set of rules is far from what

the phrase might conjure up. The term “no net loss of wetlands” also appears in the Water Resources Development Act of later that year, but in that instance it appears only as a goal, not as a concrete criterion for regulation. It is the earlier usage in the Federal Register that most analysis refers to.

Section 404 of the Clean Water Act requires anyone who wants to “discharge dredged or fill material” into a wetland to obtain a permit, which would be issued jointly by the EPA and Army Corps of Engineers (ACE). This permit is commonly known as a Section 404 permit. As usual, a lengthy and convoluted set of rules has been laid out to determine whether a permit is needed and then what specific actions will be permitted. Under certain conditions, a permit to “destroy” a wetland might be granted provided the permittee undertakes to restore, enhance, create or preserve wetlands elsewhere. This is the no-net-loss rule. We use “destroy” in a generic sense, meaning merely that the action requires wetlands mitigation.

Because of this “trading” – wetlands acres and ecological functions are surrendered in one place and restored elsewhere – the Federal government appears to be using a market in wetlands acres to govern wetland management, that is, a cap-and-trade system.

The Section 404 permits that may potentially give rise to wetlands mitigation banking are applicable to only about 15 percent of wetlands destruction in the U.S. (National Research Council, p. 72, citing EPA and ACE sources.) Agriculture and silvicultural activities are not subject to Section 404, although they may be subject to “Swampbuster” rules, described below. Even for urban and rural development, the activities that are subject to Section 404, many projects are covered by general permits that authorize activities with little agency oversight. These activities are supposed to cause only minimal environmental impacts.

Activities that are required to have a Section 404 permit must go through a three stage process. These stages are key to the performance of wetlands cap-and-trade. First, the entity applying for the permit, whom we will call the permittee, must show that he has taken steps to avoid filling the wetland; that is, he must show that there is no less environmentally damaging practicable alternative. Second, if the

action passes this test, the permittee must show that all unavoidable impacts will be minimized to the extent possible. The third stage is to compensate for any remaining impacts by restoration of wetlands elsewhere, the putative market stage.

4.9.2 The market: Wetlands mitigation banks and fees

The third stage of compensation is governed by a number of market-like set-ups.¹⁸ A permittee who is likely to apply for numerous permits over time, such as a large land developer or a state Department of Transportation, may undertake early on a single large wetlands restoration project, which then operates as a bank. That is, the wetlands compensation is undertaken *ex ante* by the same entity that will later destroy wetlands.

A natural extension of this arrangement is for an entrepreneur to undertake the wetlands restoration and then sell credits to future permittees, a procedure called commercial wetlands banking. Because the demand for future credits is highly uncertain, a commercial wetlands bank is a highly risky investment. To shift some of this risk-bearing, the EPA and ACE may allow banks to sell credits before their wetlands restoration projects are completed and certified, although the bank must provide financially-backed assurance to the regulator that certifiable wetlands will eventually be created.

A third arrangement, especially desirable for small permittees and for situations where commercial wetlands mitigation banks are not operating, is for the permittee to pay an in-lieu fee to a third party, such as the Nature Conservancy. The third party then produces wetlands credits. The in-lieu fee program provides essentially the same market-making service as a commercial mitigation bank but at a lower cost, since the fee-based provider does not need to provide the same level of financial assurance to the regulator. It is unclear how the credits are priced in in-lieu fee programs.

A fourth innovation described by Shabman and Scodari (2004) is called credit resale. Essentially, the regulatory agency acts as the market-maker, by both purchasing and selling of credits. In a sense,

¹⁸This discussion draws greatly on Shabman and Scodari (2004).

the regulatory agency bears the risks involved in matching demand and supply. The agency may be particularly suited to bearing this risk since it has the clearest picture of the likely demand for wetlands credits.

To obtain the credits that it will then sell, the regulatory agency solicits bids, using a competitive bidding process. Two important aspects of this process that determine its efficiency are (i) the procedure the agency uses to solicit the bids and (ii) the relationship between bid prices and the prices that the credit purchasers must pay.

Note that many of these market-making institutions are similar to ones used in the context of water quality trading to finance nutrient reduction by non-point sources.

4.9.3 Economics of no-net-loss: The cost-effectiveness of wetlands preservation

Shabman and Scodari (2004) claim that the full set of rules creates an unwieldy regulation that “should not be viewed as an application of a market-like environmental policy.” This section argues that the no net loss rule does give rise to a market in wetlands credits, albeit a thin and possibly inefficient one. The market is only a small part of the way in which wetlands protection is governed. A large part of wetlands protection remains governed by command-and-control regulations.

The desirable trait shared by market-based approaches is that the environmental service, whatever it might eventually be, is provided at least cost. This is the standard against which wetlands policy can be judged. We see four aspects that affect the cost-effectiveness of the non-net-loss provisions.

First, the no-net-loss provision and the accompanying trading come at the end of a long process involving multiple decisions about what kind of activities will be permitted. This is an information-laden command-and-control process, not a market one. There is no guarantee that efficient or cost-effective decisions will result. The authorizing legislation does not easily lend itself to market-based approaches.

Second, conditional on a permit application making it to the third stage, the trading institutions appear relatively efficient. These institutions may be judged both in terms of their current efficiency, which is

difficult to measure, and the ability of the agencies to improve on perceived inefficiencies. Note that there are many different ways to “make” a market – many different legal and regulatory arrangements that can facilitate trading. Although most of the currently used arrangements appear reasonable, it is difficult to compare the efficiency properties of these institutions for wetlands trading because of the thinness of the market and idiosyncratic risks involved in wetlands mitigation. Note also that the agencies involved, mainly EPA and ACE, appear to have shown some creativity and flexibility in developing viable institutions. At least four mechanisms have been used, and we expect further innovation in the future. Such innovation strikes us as exemplary, even though the evolution of these set-ups points out the flaws in each.

Third, there is a great deal of uncertainty in terms of regulatory approval and timing. Credit producers face highly uncertain demand because they do not know how many projects, and of what size, will make it to the third stage, at which point they become potential credit purchasers. They also face some risk in knowing whether their projects will be certified. These risks, and the way they are borne by the various parties, add to the difficulty of evaluating institutions for wetlands trading. It is not clear who should optimally bear this risk.

Four, wetlands provide multiple services. The no-net-loss goal is stated in terms of overall wetland functions; however, wetland area by wetland type is often used as a proxy for “function.” Building a trading system that focuses explicitly on wetland functions is conceptually possible through the use of a weighting scheme to add up the various functions, but this is difficult in practice. Without such a system, rules that are cost-effective in terms of area (measured in stage-three acres), which we believe this policy achieves, need not be cost-effective in terms of wetland functions.

A similar argument applies to where the restored wetlands are located. Restoration of wetlands in the same watershed is typically preferable to restoration of wetlands in some other watershed. But, as with all location-based environmental credits, any move to restrict the market to local trades results in a thin, perhaps even non-existent market.

The efficiency of wetland banking and mitigation is conditional on the third “compensation” stage being reached. It is unlikely that an efficient set of actions is being undertaken in the first two stages, given the highly prescriptive nature of the regulations. Such prescriptive regulations are required, however, by the authorizing legislation.

4.9.4 Other market-oriented habitat policies

4.9.4.1 Swampbuster

Agricultural activities are exempt from Section 404 of the Clean Water Act. Some of these activities are regulated instead by the Food Security Act of 1985 and further modifications in the 1996 Farm Bill. The so-called Swampbuster provisions specify that a landowner who plants crops in converted wetlands will not be eligible for agricultural support programs unless he or she undertakes mitigation (that is, restoration, enhancement, creation, or preservation of wetlands) on a one-to-one basis. Thus, this provision has the same implicit cap-and-trade feature as the no-net-loss rule.

4.9.4.2 Habitat conservation under the Endangered Species Act

The Endangered Species Act of 1973 prohibits any activity that might harm a species listed as threatened or endangered. A series of court decisions over the years have interpreted “harm” to include significant habitat modification. A later amendment to the ESA allows what is called an “incidental take,” in which activities might cause such habitat modification but the modification is not deemed to threaten the survival of the species. Such activities require a permit, called an incidental take permit.

One of the requirements of such a permit is the filing of a Habitat Conservation Plan (HCP). The purpose of the HCP is to show how the permittee will minimize the effects of the activity and mitigate any of the effects that remain. Thus, the policy is basically the same as for Section 404 permits, although the regulatory structure (and regulatory authority) is different. One of the actions that may count as mitigation

is the restoration, enhancement, or preservation of wildlife habitat elsewhere. This provision looks like it opens the door to trading of what might be labeled “habitat credits.”

Although the procedure is similar to the no-net-loss rule, the market is much more rudimentary and lacks the institutions that wetlands credit markets have. The market appears to be extremely thin. This thinness is not surprising. Opportunities to enhance the habitat of endangered species are highly limited and not nearly as interchangeable as for wetlands. With endangered species, each species has separate habitat needs. Although commercial “habitat banks” are conceptually possible (although the legal precedents are much murkier than for wetlands banking), they would have few options for developing suitable credits. Predicting demand is difficult. Furthermore, the same feature that makes suitable habitat scarce also means that the demand for habitat credits is even more uncertain than for wetlands credits.

4.10 Tradable development rights

4.10.1 Overview

State and county governments in the U.S. have established a wide variety of programs to slow the conversion of farmland and other open space to developed uses. A small number of these programs use tradable development rights, also known as TDRs. TDR programs exist in roughly 50 jurisdictions (counties, towns, and townships), although their size and success are highly asymmetric. According to the American Farmland Trust (2001), only 15 programs nationwide have protected more than 100 acres. Montgomery County, Maryland, the most ambitious of the TDR programs, has protected roughly 60 percent of the national TDR total (American Farmland Trust, 2001).

The TDR concept is also used to regulate building heights in several major U.S. cities, having started in the mid-1970s (Levinson, 1997). Thus, TDRs join the Clean Air Act’s offset program and CAFE standards as part of a spurt of market-oriented policies around that time period. The intellectual foundation appears to have been laid by Costonis (1973). The first economic analysis in the academic literature is due to Field and Conrad (1975).

TDR programs, which are usually administered on a county-level basis, work in conjunction with the jurisdiction's baseline zoning rules. Zoning in the U.S. is carried out at the county level and specifies, among other things, the permissible density of housing in each area of the county, among those areas deemed suitable for residential use. Any county that wishes to institute a TDR program must simultaneously define a baseline zoning plan (sometimes known generically as a Master Plan), which lays out allowable densities and draws the boundaries that pertain to each allowable density.

The TDR program then specifies some areas as eligible to increase their densities above this baseline level. These are called receiving areas. At the same time, the TDR program specifies other areas as eligible to sell TDRs to the developers. These areas are known as sending areas. Areas can be classified as receiving, sending, neither, or both.

A developer who wants to increase density in a receiving area must buy TDRs to cover the increased density. TDRs are purchased from landowners in the sending area. Any landowner with parcels in the sending area that are below the baseline zoning is eligible to create and then sell a TDR. A landowner in a sending area creates TDRs by permanently agreeing not to develop the land beyond its current density. The number of TDRs created depends on the difference between the current density and the baseline zoning. If the landowner sells his or her TDRs, then the land cannot be developed beyond its current density.

The framework is similar to point-non-point trading, in that the cap falls primarily on the urban receiving area. The regulated urban developers must find willing sellers in the rural sending areas and negotiate with them the selling price for their TDRs.

Note that some landowners may not be in designated sending *or* receiving areas. Such landowners are subject to the baseline zoning rule and cannot exceed the allowable density (even if they were to purchase a TDR) nor sell TDRs if they decide not to develop their property.

TDR programs exist within a multitude of other zoning rules and other programs to control farmland conversion. TDR programs differ across jurisdictions and across time in the baseline zoning rules, including the rights to appeal zoning rulings; restrictions on development in receiving areas conditional on the purchase of TDRs; and rules

concerning the allowable development on a parcel that has sold its development rights. This latter is, in essence, the definition of a development right, and may differ from jurisdiction to jurisdiction.

TDR programs often exist in conjunction with state- or county-financed programs to purchase development rights. These are known as purchase-of-development-rights (PDR) programs. PDR programs may be administered in the absence of a TDR programs and are much more common than TDR programs. Under a PDR program, the state or county (or non-profit organization) acts as a buyer for development rights, but those rights are not used to increase density elsewhere in the county.

4.10.2 Example

Because trading of development rights may be less familiar to readers than other trading, this section describes the TDR rules in place for Calvert County, Maryland. Calvert County's TDR program has been in place since 1978. The first development right was transferred in 1981.

The TDR program starts with the baseline zoning rule. In Calvert County, current baseline zoning density (per 10 acres) is 1 unit for agricultural areas and rural communities, 5 units for non-town residential areas, and 20 units for towns. A unit is "one residence," roughly speaking. A unit has a precise legal definition that is beyond the scope of this article and will, of course, differ across programs.

Designation of parcels into these four categories is also made by the county's zoning board. Note that zoning rules are subject to change. Of course, once development has taken place it is never required to be undone if the zoning rule changes.

TDRs may be sold only by those landowners in agricultural areas and rural communities; these are the sending areas. Areas outside these zones cannot sell TDRs even if they commit to staying below the baseline density.

The allowable densities, conditional on the developer having purchased the requisite number of TDRs, are 2 units (per 10 acres) for agricultural areas, 5 units for rural communities, 40 units for non-town residential areas, and 140 units for towns.

Thus, a rural landowner who owns 100 acres with only one dwelling has three options: (i) She may develop up to 10 units, since the baseline zoning is 1 unit per 10 acres. (ii) She may sell 100 TDRs and then develop none of her own acres. Under current rules, she may keep the existing house and may build one additional house that can only be occupied by family members. This landowner may sell her house and its acreage at any time and to anyone, but the new owner and all subsequent owners are subject to the restriction against further building. (iii) She may purchase up to 10 TDRs and then develop up to 20 housing units at 2 units per 10 acres. Note that many TDR programs do not allow landowners in sending areas to use TDRs to increase densities above the baseline density.

The purchaser of 100 TDRs has multiple options as well. He may build 100 units in any of the receiving areas, at densities above the baseline zoning level but below the maximum.

Consider the TDR requirements for development in a town center. If the parcel is currently at the baseline zoning of 20 units per 10 acres, then up to 120 more units can be built, conditional on the developer holding the requisite TDRs. If the parcel is currently below the baseline zoning, say completely undeveloped, then the builder can build 120 units using only 100 TDRs, or 140 units contingent on the purchase of 120 TDRs.

4.10.3 Assessment

The success of TDR programs is difficult to judge because of the small number of active programs, the complexity of land uses, and difficulties in establishing a counterfactual. Our assessment is based on conceptual strengths and weaknesses.

4.10.3.1 Strengths

The TDR program shares many similarities with water quality trading with non-point sources. Like the water pollution case, one of the key strengths of TDRs is the harnessing of both private financing and entrepreneurship, in this instance for farmland preservation (as in Section 4.5.4.) Under the alternative PDR program, states and counties

pay for development rights directly. The TDR program relies on private developers to finance the purchase of development rights.

Both TDR and PDR programs redistribute the costs of farmland preservation in a way that benefits rural landowners, compared to direct down-zoning of rural areas. Down-zoning means the reduction in the allowable density in rural areas, which is undertaken by county governments as a further tool to preserve farmland. TDR programs often are adopted in conjunction with down-zoning (McConnell et al., 2005). Down-zoning reduces land values for rural landowners who were planning on developing their parcels. By simultaneously introducing a TDR program, the county gives these landowners something of value, the ability to sell TDRs. (The TDR program does not completely neutralize the distribution of the costs of down-zoning.) Note that this feature is similar to what occurs under pollution cap-and-trade when initial allowances are given to existing polluters rather than auctioned. An initial allocation makes these entities less likely to oppose the down-zoning, in the case of TDRs, or the lowered emissions in the case of pollution cap-and-trade.

Like non-point source trading, the TDR program also harnesses private entrepreneurship in finding suitable properties whose landowners may be willing to sell their development rights at the lowest price. We argue in Section 4.10.3.2 that this entrepreneurship does not, however, achieve cost-effective farmland preservation.

TDRs create more successful markets than does water quality trading. The reason is that individual trades do not need the same lengthy process for approval. The *ex ante* specification of allowable trades, which would be a desirable feature of water quality trading but is difficult to accomplish, is a straightforward feature of TDR programs. Therefore, trades and price negotiation can be carried out under certainty and without much regulator oversight. Transactions costs are still not trivial, however. The sale of a TDR is a complicated legal transaction due to the placement of a restriction on the sending-property's deed. In some programs, a developer needs approval to use the TDRs on the receiving plot.

4.10.3.2 Weaknesses

The key apparent weakness of TDR programs is that many of the TDRs that are sold are from properties that would be highly unlikely to be developed. Therefore, the TDR program increases density in urban areas (where the purchased TDRs are used) without reducing density in rural areas. This problem has also been pointed out in a conceptual model of floor-area-ratio trades (essentially, building height trading) by Levinson (1997).

The TDR programs result in the preservation of low-development-probability parcels because these parcels have the lowest value for their development rights. The development right's reservation value for parcel that is unlikely to be developed is close to zero. Purchasers of TDRs end up buying these TDRs because the purchasers have an incentive to search out the lowest-priced TDRs. Note that PDR programs do not suffer to the same extent because the state, county, or non-profit purchaser can choose to purchase TDRs from properties that are believed to have a higher probability of development.

The magnitude of this problem could presumably be estimated. Qualitative evidence is available from Montgomery County, where most of the TDR sales have come from parcels on the edge of the county, the farthest point from Washington DC (Horowitz and Lynch, 1998). The probability of development is roughly inversely proportional to distance from a major urban center.

It is conceptually possible to solve this problem by defining equivalence ratios that compensate for differences in the probability of development across the sending area. No TDR program has incorporated this option.

A second problem that has arisen in TDR programs is that increased density in receiving areas is also an externality to homeowners adjacent to areas that are being "super-developed" using TDRs. The exact nature of this externality can be difficult to pin down, but at least some landowners assert that their quality of life (or even property values) is diminished by the establishment of the higher-density housing made possible by the purchase of TDRs.

TDR programs are necessarily carried out at local levels. Farmland conversion may therefore be shifted to other counties rather than slowed in the state as a whole. This problem exists even within the county. Many rural areas in a county may currently be below the allowable density. As properties are preserved, the remaining ones face increased pressure to develop up to the allowable density. In the extreme, there may be no effect on county-level farmland conversion.

A final point on the efficiency of TDR programs is that they are laid on top of existing zoning rules pertaining to baseline densities and area designations. These rules may not be cost-effective or efficient. Therefore, a TDR program may (if designed properly, say with appropriate equivalency ratios) be cost-effective in county-level farmland preservation, but the overall system of rules-plus-TDR-program may not be.

4.11 Individual transferable quotas for fisheries

4.11.1 Overview

Fisheries have traditionally been open access resources, subject to a true Tragedy of the Commons. In the absence of regulatory remedy, fishermen individually have no incentive to limit their fishing activity, since any fish that get away would likely be caught by someone else rather than remain at large to breed and contribute to the fish stock. The result of these aggregate actions is depletion of fish stocks. Governments and other entities have therefore tried to limit the amount of fish caught in any given year. Indirect methods of limiting the catch include restrictions on the type of fishing gear or the size or number of fishing vessels. These have almost always been ineffective because fishermen adopt other fishing strategies, with little subsequent reduction in the catch. Direct methods of limiting the catch, such as closing the season after a certain aggregate volume are caught, are imprecise and the resulting fishing methods highly inefficient.

Individual transferable quotas (ITQs) are a cap-and-trade program for fisheries designed to overcome these problems. The first major U.S. ITQ program was developed in 1989 for surf clams and ocean quahogs (a type of clam), although it had its roots in the 1976 Magnusson Act, which authorized federal agencies to begin to limit fishing effort.

McCay (2001) notes that the idea of “stock certificates” (albeit presumably non-tradable) dates to 1978, the time period in which many other cap-and-trade programs were being discussed. The first theoretical discussion in the academic literature is due to Anderson and Hill (1975).

There are currently federal ITQ programs for three fisheries: surf clams and ocean quahog, Alaska halibut and sablefish (technically referred to as an individual fishing quota (IFQ) system), and Bering Sea crab. For simplicity, we will refer to these as clam, halibut, and crab ITQ programs. A fourth federal ITQ program for wreckfish, once active, has been disbanded, apparently because of low fish prices rather than a failure of the ITQ. Florida operates a state program for spiny lobster; a few other state programs also exist. Maryland operates a permit-based program for rockfish but does not allow trading. Permit-based programs without trading are similar to many (state) hunting regulations in which a fixed number of permits, each corresponding to the right to shoot one animal, are issued each year, often through a lottery. A small number of state hunting programs allow permits to be traded (e.g., deer hunting in Kansas.)

The ITQ system operates by defining a quota, which is a share of the annual total allowable catch (TAC). Prior to the beginning of the relevant fishing season, the fishery management council that oversees each of these ITQ programs decides the total allowable catch (TAC) for that fishery for the year, denominated in either volume or pounds. Quota shares are then translated into tonnage or bushel limits for that year, called allocation permits. Under the ITQ system, fishermen are required to hold allocation permits equal to the amount of fish they catch. Thus, the ITQ system curtails overfishing by limiting the total catch. It prevents inefficient fishing effort by guaranteeing each permit holder the opportunity to catch his fish in the most efficient manner.

The quota shares are long-lived assets that may be sold or leased. In general, the yearly allocation permits cannot be banked or borrowed, although they may be used at any time throughout the fishing season. Because of unpredictable under- or over-harvests, the halibut program allows up to 10 percent of the TAC to be carried over to or borrowed from the next year’s allocation.

Adherence to the permits is enforced in a variety of ways. Surf clams and ocean quahogs are subject to dockside inspection and cross-checking of logbooks between vessels and processors (NRC, 1999). Clams are sold in bushel-size cages, each of which requires a tag, which is the physical manifestation of an allocation permit. Halibut landings must be delivered to registered buyers; these landings are subject to real-time monitoring using an IFQ Land Card, essentially a form of debit card. Larger halibut vessels may have an observer on board. All fisheries are also subject to aerial and at-sea surveillance, checking of logbooks, and similar traditional monitoring and enforcement actions.

ITQs are a response primarily to inefficient fishery regulation, although they also are useful in building up fish stocks. Prior to their ITQ programs, these fisheries were managed through a variety of means, each spectacularly inefficient. Surf clams were an open access fishery prior to 1978. In 1978, a moratorium was issued for new vessels, a restriction that did not curtail overfishing and that led to older vessels remaining in the fishery when they might optimally have exited. Fisheries have also been governed by gear restrictions, limits on the number of trips, or limits on catch per trip.

In most cases, a TAC was also imposed on the fishery on top of these restrictions (with no individual permits) with all fishing effort being prohibited after the TAC was reached. This approach typically leads to a race in which many extremely large vessels concentrate their fishing effort in a very short time period. To enforce the TAC (or in the absence of true TAC controls), the fishing season may be sharply curtailed. Prior to the IFQ introduction, the halibut fishery was sometimes open for just 24 hours per year (Pautzke and Oliver, 1997); the surf clam fishery was sometimes open for just 6 hours every other week (Buck, 1995). The analogy to hunting is again apt: in 2005, Maryland's black bear season was closed just a few hours after it opened.

This regulatory approach is inefficient because it leads to overcapitalization of both the fishing fleet and the processing industry, which must process a large volume of fish in a short period of time. The race may also have led to higher accident rates, dumping of unwanted catch, and abandoned gear (NRC, 1999).

The clam and halibut ITQ programs have largely managed to overcome these problems. Because fishermen have a fixed right to a share of the catch, they can allocate fishing effort throughout the season, which allows both vessel owners and processors to select an efficient size for their operation. The ability of vessel owners to manage the timing appears to be the most frequently cited benefit of the ITQ approach. Gear restrictions, trip limits, and similar restrictions have been mostly but not entirely eliminated (Buck, 1995).

Fishery permits are also mostly freely tradable, although there are some restrictions. Quota shares for clams cannot be traded during the last two months of the season (Buck, 1995). The halibut fishery is divided into areas and vessel-size classes; permits cannot be traded across areas or vessel-sizes. The purpose is to ensure that small vessels are not driven out by larger vessels. In the halibut fishery, entrants (owners who were not eligible to receive an initial allocation of quotas) must be onboard the vessel when the shares are caught. Limits on the number of quotas that can be owned by one entity are also a prominent feature of the halibut fishery, although this restriction appears to be being phased out. The halibut fishery also allows no more than 10 percent of the shares to be leased in any one year, for some categories of vessels.

Although the clam, wreckfish, and halibut ITQ programs were considered successful by most accounts (e.g., NRC, 1999), Congress in 1996 placed a ban on the introduction of any new ITQ systems. The concern was the possibility of market power by a small number of quota owners and the exit of small, community-based fishermen from the industry, with an accompanying effect on the vitality of fishing communities. The ban was lifted in 2002 and the crab fishery ITQ-type system was introduced shortly thereafter.

4.11.2 Quota allocation and the TAC

The ITQ approach to fishery management typically involves a one-time decision over the initial quota allocation and a yearly decision over the total allowable catch. Most ITQ fisheries have opted to mimic the status quo as much as possible in the allocation of initial quotas, using various

formulas based on historic catch and, to a lesser extent, vessel size. This decision has been useful in achieving political acceptability (McCay, 2001). A yearly auctioning of the quotas appears to be prohibited under the authorizing legislation.

Most economists would predict that initial allocations would have no effect on the long-run status of the fishery. (Weisman, 1997, cited by McCay, 2001) found that the larger the initial allocation of surf clam and quahog quotas, the greater the likelihood that a firm would still be in the fishery four years later, although the size and statistical significance of the effect were not large.

The setting of the TAC has much more substantial consequences. TACs are set by regional fishery management councils based on voting by members, which include both quota owners, government administrators, and academics. Uncertainty over the size of the stock makes TAC decisions difficult and contentious. Turner and Weninger (2005) analyze TAC-setting for surf clams and show that quota owners, who have a long-run stake in the fishery, are more likely to want lower TACs than non-quota owning fishermen, who do not have the same long-run stakes.

4.11.3 Assessment

ITQ programs represent a transparent and mostly faithful application of cap-and-trade principles. The inefficiencies of previous regulations, including the race to capture each year's total allowable catch, were easily observed, so administrators were amenable to dropping such regulations and relying on the quota system to regulate the catch. Trading has been able to proceed with minimal oversight. These attributes suggest that many ITQ programs will have come close to the full cost-effectiveness available from cap-and-trade.

Because ITQs allow the fishing season to be extended, consumers may also be affected, a benefit that has not received analytical attention. If consumers have decreasing marginal values for fish, a more even supply of fresh fish leads to higher consumer surplus under the ITQ system than under the open-access TAC system.

Some restrictions on trades remain. These have been used to favor smaller vessels and to slow the inevitable transition of the fisheries. If there is a social value to keeping small operators in the industry, this restriction may not reduce efficiency. New rules may allow communities, rather than individual vessel owners, to hold quotas.

Note that the current set of ITQ programs require quota owners to be active fishermen; non-fishing individuals such as environmental groups are prohibited from purchasing or leasing quotas. This restriction may have an efficiency cost.

Other problems occasionally linked to ITQs are high-grading, by-catch, and poaching or cheating. High-grading means the disposal of smaller or less desirable fish that would otherwise require an ITQ. By-catch in this context means the disposal of other species (*i.e.*, those not requiring an ITQ) so that the vessel can fill up with the desired species. Buck (1995) argues that these problems are no worse under ITQs than under open-access and are likely actually alleviated by the ITQ approach because quotas allow vessels to choose efficient fishing technology and timing. Although poaching problems are also raised, we have not found empirical or anecdotal evidence of its occurrence.

An ITQ program can be more than cost-effective; it can be efficient if the TAC is set at the efficient level.¹⁹ The ITQ program appears to have made efficient setting of the TAC more likely, a second benefit that seems to have been unanticipated. Because the quotas have given owners a long-run stake in the fisheries, they are willing to move closer to a long-run equilibrium stock. In several years, the TAC for surf clams has not been binding (Turner and Weninger, 2005).

When the moratorium on new ITQs was passed in 1996, Congress also directed the National Academy of Sciences to study the efficacy of these regulations, both in the U.S. and elsewhere. The result was a report that strongly favored the ITQ approach.

Thus, it is disappointing that the newest ITQ program, for Bering Sea crabs (awkwardly called the Crab Rationalization Program),

¹⁹It is unclear from a conceptual standpoint whether taxes or permits, the two alternative methods of cost-effective regulation, would have created the same strong incentives for efficient setting of the TAC in a public choice framework.

appears not to exhibit the successful properties of the other ITQs. The crab fishery has two separate permit systems, one for fishing vessels and one for processors. The Individual Processor Quota (IPQ) system grants market power to processors in both the purchase of the raw product and in sales to consumers. Note that no open-access problem occurs for the processing industry. The IPQ system therefore should likely be considered an “anti-market” approach.

Finally, we again find that the market-based ITQ system has been laid on top of an existing, potentially inefficient regulatory structure rather than replacing it. Halibut fishing must be done with long-lines rather than fishermen being free to choose catch technique. In the crab case, vessels are restricted to 450 crab pots per vessel. The efficiency costs of these regulations have not been estimated.

4.12 Greenhouse gas emissions

Most proponents of regulating U.S. greenhouse gas emissions – whether as part of an international agreement such as the Kyoto Protocol or as a purely domestic approach – have advocated the use of a market approach. Such a market approach would consist either of a carbon tax (more precisely, a carbon-dioxide tax) or a cap-and-trade system. Like the other examples in this article, this section does not address the broader question of why the U.S. might choose to reduce greenhouse gas emissions or how stringent such regulation should be.

Greenhouse gases seem particularly suitable for a market approach. Emissions can be easily (albeit indirectly) monitored and come from a sufficient number of sources that an allowance market, if cap-and-trade is selected, would be competitive. Almost all U.S. sources could be covered, although carbon from vegetation and di-nitrogen oxide from soils present some challenges. The damages from emissions are also independent of where emissions originate, so hot spots or downwind problems do not arise. The cost-savings from a market approach would be substantial, since command-and-control approaches, even if they took the relatively benign form of performance standards (for example, energy efficiency standards for new automobiles, buildings, and appliances), would be grossly inefficient.

4.12.1 Market-based regulation of greenhouse gases

Because greenhouse gas emissions are near-perfectly proportional to the quantity of fossil fuels combusted, a carbon cap-and-trade or carbon tax can be based on the quantity of fossil fuels used as energy sources, with equivalence ratios based on the carbon content of each fuel. Thus, while emissions may not be able to be directly monitored, they can be inferred by the fuel inputs used by various sources. This approach requires that fuel use be monitored, either on the demand or on the supply side. Non-combustion uses of fossil fuels such as fugitive methane or non-fossil-fuel sources such as carbon from deforestation or N_2O from soils are less easily accommodated in a market approach, although the market approach might be extended even to these sources. The U.S. is not a substantial source of black soot.

Consider first the design of a domestic carbon cap-and-trade system for fossil fuels. Economists recognize two types of carbon cap-and-trade which differ according to who is required to hold the allowances.²⁰ Under an *upstream* trading system, producers and importers would be required to hold allowances for the quantity of fuel they produce or import, based on the carbon emissions of their fuel when it is eventually combusted by a user. Under this system, individual consumers and businesses would not hold allowances and would, in essence, have no contact with the allowance system. Those consumers would nonetheless have an incentive to reduce fuel use because the allowance cap, which would be set below the current level of carbon emissions, would lead the prices of fuels to rise above their non-capped levels. Consumers and businesses and other downstream energy users would then have an incentive to reduce energy use when that energy came from fossil fuels. As with other market instruments, consumers would have complete flexibility to choose how best to respond to higher prices. This flexibility would lead to carbon emission reductions being achieved at the lowest cost to the economy.

The alternative is a *downstream* trading system, under which energy consumers would be required to hold the allowances. One possibility

²⁰This section draws heavily on CBO (2001).

is a system in which individual households and businesses would be required to hold allowances that they would submit when paying electric and gas bills or buying gasoline at the pump. Downstream trading systems are instead usually proposed to cover a more manageable number of “large” sources of emissions, such as electric utilities, large manufacturers, government facilities, commercial transportation fleets, and some large employers. This latter approach would therefore not cover all sources of greenhouse gas emissions.

Under an upstream system, approximately 2,000 sources would need to be monitored for fuel use and allowance holdings (CBO, 2001; Cramton and Kerr, 1997). The number of sources that would need to be monitored under a downstream system depends on the coverage. A downstream trading system that covered all users, including individual households, would require monitoring the fuel consumption of millions of agents; such monitoring could be made simpler through debit cards and electronic tracking of transactions but administration would still be a substantial and intrusive undertaking. A downstream system that covered all users would have identical economic effects to an upstream trading system but would be more costly to implement due to the higher administrative costs. A downstream trading system that covered only “large” users would be less expensive to implement than a full downstream system but would not achieve emission cuts at lowest cost. It would also present the potential problem of emissions “leaking” from regulated to unregulated sources. An upstream system therefore is clearly superior to any form of downstream system.

These issues apply equally to a carbon tax, although existing rules for taxation of fossil fuels (unrelated to carbon emissions) would likely dictate at which level the tax would be levied.

The U.S. Congress has considered two major cap-and-trade proposals for greenhouse gases. The proposed Clean Air Planning Act of 2003 would have imposed a carbon cap on electric utilities, a form of downstream trading system that would cover roughly 37 percent of U.S. emissions. The proposed Climate Stewardship Act of 2003 would have imposed a downstream trading system on the “electricity, transportation, industry, and commercial sectors.” Thus, the precise coverage has not been spelled out, although it is clear that a downstream system

is envisioned that would be larger than that covered by the Clean Air Planning Act but still short of a full system. A true upstream system has not yet been seriously considered by Congress. California has also recently discussed its own statewide cap-and-trade.

It is worth noting that these legislative proposals have not been accompanied by proposed removal of CAFE standards, energy efficiency standards, or other energy efficiency regulations. Estimated costs of proposed systems often do not make clear the assumptions about other energy regulations and do not estimate the added cost from such regulations. Many of the existing energy-efficiency regulations may be presumed to become non-binding under a carbon cap or carbon tax. However, it is also possible that cap-and-trade would be accompanied by, say, a tightening of CAFE or other energy efficiency standards; in this situation, these regulations would add additional costs but no additional benefits.

As discussed in Section 3, there are many design features that can reduce the cost or otherwise improve the performance or appeal of a cap-and-trade system. An upstream cap-and-trade program for greenhouse gas emissions should include provisions for: (i) banking and borrowing, which are clearly warranted due to the stock pollutant nature of greenhouse gases; (ii) a safety valve to hedge against the potentially high cost of regulation; and (iii) auctioning rather than grandfathering of the allowances. The auctioning of allowances, with the revenues used to offset other inefficient taxes or to decrease the federal deficit, could be particularly valuable due to the high aggregate value of the allowances.

Although fossil fuels account for a large proportion of U.S. greenhouse gas emissions, increased carbon sequestration in soils and plant cover represents another option for reducing emissions. The costs of carbon sequestration are predicted to be low (on the margin); thus, this issue is similar to water quality trading, although with the difference that non-point sources of carbon emissions are a smaller proportion of overall emissions. As with non-point water quality trading, opt-in problems would arise.

A carbon cap-and-trade program could incorporate sequestration more readily than a carbon tax approach. A domestic cap-and-trade

system would also mesh with an international market for carbon credits in a way that a carbon tax would not.

4.13 Market regulation of lead in gasoline and ozone-depleting chemicals

Two successful yet less widely cited cap-and-trade policies are for lead in gasoline and ozone-depleting chemicals. Both programs are now defunct because the substances they controlled have been banned.

4.13.1 Lead

Lead was added to gasoline starting in the early days of the automobile. Airborne lead, however, causes serious health problems including lowered IQs. Lead in gasoline also ruins catalytic converters, which were required on new cars starting in 1975 to control emissions of hydrocarbons, nitrous oxides, and carbon monoxide. This latter problem led EPA to require cars equipped with such converters to use unleaded gasoline. Increasing concern about adverse health effects from airborne lead further led EPA to sharply curtail lead levels in the remaining leaded gasoline market.

The EPA's first response was to reduce allowable lead concentrations (averaged quarterly at the refinery level over both leaded and unleaded gasoline) from 1.7 grams per gallon to 1.0 grams per gallon, to be phased in from 1975 to 1979. This policy allowed intra-refinery trading but not banking or inter-refinery trading. Later, some trading among refineries owned by the same company was permitted. In 1982, partly in response to heightened awareness of lead's health consequences, EPA shifted to regulating the lead content of leaded gasoline. The allowable lead concentration (averaged quarterly over leaded gasoline only) was reduced from 1.1 gram per gallon to 0.1 grams gallon, to be phased in from 1982 to 1987. This second and better known policy allowed both banking and inter-refinery trading. In 1988, EPA shifted back to an allowable lead concentration of 0.1 grams per gallon to be met at the individual refinery level, again without banking or inter-refinery trading. Lead was banned as a gasoline additive starting in 1996.

The lead program was considered a success. Numerous trades were made, which suggests, although does not prove, that trading provided substantial cost savings over a pure performance standard. Kerr and Newell (2003) argue that adoption of new technologies was stronger under the trading regime (1983–1987) than the performance standard regimes. The program had its drawbacks, however. Kerr (1993) notes that over half of the trades (by volume) were between refineries owned by the same company.

As with other market mechanisms, regulators did not entirely relinquish control. Small and very small refineries faced less stringent standards until 1983 (Kerr and Newell, 2003, p. 321). Also, after a period of successful trading, EPA removed inter-refinery trading and re-imposed a performance standard.

The lead program is economically similar to Renewable Portfolio Standards. Like CAFE or RPS, no “initial allowance” decision is needed for a cap-and-trade system when a rate or average is being capped. This feature smoothes the introduction of a cap-and-trade. The lead program, again like CAFE and RPS, is actually more like a tax than a true cap-and-trade (see Section 4.8.2.3) but uses the concepts and vocabulary of cap-and-trade.

The low volume of inter-firm trading that appears to have plagued the lead program seems unlikely to manifest itself under RPS given that regulators and sources, especially electric utilities, are much more familiar now with credit trading. On the other hand, renewable electricity is more complicated to monitor than lead in gasoline, which may contribute to the regulatory uncertainty that appears to be plaguing RPS.

4.13.2 Ozone depleting chemicals

In 1987, the Montreal Protocol established country-specific limits on the use of ozone depleting chemicals (ODC, sometimes also called ozone depleting substances). To meet the U.S. obligation, EPA set up a domestic cap-and-trade program using the *upstream* format. Yearly allowances were distributed for free to 8 producers and 20 importers based on their 1986 market shares and the overall cap (EPA, 2001). The cap declined through time. Class I substances (and the

ODC trading system along with them) were completely phased out in 2000, with the exception of methyl bromide and some essential use exemptions.

The trading system had several innovations. Title VI of the 1990 Clean Air Act Amendments authorized trading among chemicals with the same ozone depletion potential. International trading was also allowed, although trades required EPA approval.

In 1989, excise taxes on ODC were applied on top of the cap-and-trade system. Tax rates were based on each chemical's ozone depleting potential, making them a possible model for possible greenhouse taxes. The tax was also applied to existing stocks of chemicals.

Because the expectation was that the cap would be binding, the tax's intended purpose was to capture the windfall that the cap gave to existing ODC producers (Hoerner, 1997), which was desirable because of Congressional efforts at that time to reduce the Federal budget deficit. A tax on ODC when the cap is binding would thus be an alternative to auctioning of initial allowances. The view of the ODC taxes as capturing windfall profits was an important element of the legislation's passage, since it could then be argued that it did not violate President Bush's no-new-taxes pledge (Hoerner, 1997). In the ODC case, however, alternative chemicals and processes were developed relatively quickly and the cap was not binding for the first four years after its implementation. This gave the tax a presumably desirable and (mostly) unanticipated role in reducing chemical use. Although it should follow that the allowance price was zero during the non-binding-cap years, this has not been shown.

The cap-and-trade-and-tax system also had its share of non-market elements, however, as usual. Banking and borrowing were not permitted. Whenever allowances were traded, one percent of the traded volume was "retired" (EPA, 2001). This retirement followed from a provision of Title VI which required that transfers be allowed only if the change in annual production "exceed[ed] the reduction otherwise applicable to the transferor." Legislators seemed not to be able to believe that cap-and-trade exhausted all of the available cost-savings.

In addition to the cap-and-trade system, individual uses of ODC were still subject to regulation, such as rules governing maintenance of

car air conditioning systems. Although such non-market elements might typically be considered undesirable, their efficiency properties are complicated here. ODC create an externality only when they escape into the air. The ODC cap-and-trade does not precisely target *escapement*, so additional rules to reduce escape may be valuable. On the other hand, escape (from air conditioning systems, for example) is inevitable over the lifetime of the chemical, so the ODC cap may indirectly target the true externality. In the absence of time-of-release considerations, the non-market rules would seem to be inefficient.

Finally, we should note that the cap-and-trade system was considered extremely successful despite these non-market features. EPA (2001) notes that administrative staffing and record-keeping costs were considerably less than would have been needed to “regulate end uses” or for a “traditional regulatory approach.” It is not clear what counterfactual regulation is being referred to in citing these apparent savings. OECD (1998) argues that international trading was also valuable: “Dow Chemical ended methyl chloroform production in Canada when the market declined and shifted 4.5 million kilograms of production to the United States. It cost Dow less to boost United States production and ship the chemical to Canada than to run two under-capacity systems” (OECD, 1998, p. 33). This is a classic example of the gains from trade.

5

Conclusions

5.1 Five conclusions

Market mechanisms occupy a central position in environmental economics research, and support for taxes and cap-and-trade to address environmental problems is one of the most consistent messages from the economics profession. This paper has attempted to outline the major results and themes of this research. We have also shown how market mechanisms are used in U.S. environmental policies.

We draw the following conclusions:

1. Cap-and-trade is preferred over taxes by U.S. policymakers. Both environmental taxes and cap-and-trade systems, the two forms of market-based regulation, are used in U.S. environmental policies but cap-and-trade systems are considerably more prevalent. They are used in several new regulations and proposed legislation pertaining to air and greenhouse gas emissions, in numerous water quality trading schemes across the U.S., in state approaches to farmland preservation, and in fisheries. Taxes or other price approaches have shown up in a significant way only in gas and hazardous waste taxes, and even here they have not been treated as the backbone of a regulatory policy.

As a further example of the primacy of cap-and-trade, consider the climate change policy that has made it farthest in the U.S. political system, the proposed Climate Stewardship Act. This Act proposed a cap-and-trade system for carbon fuels. Greenhouse gas emissions from carbon fuels arguably could have been tackled more simply, more quickly, and with less regulatory uncertainty using carbon taxes since taxes would use an already-existing institutional and legal infrastructure. The alternative virtues of cap-and-trade, however, whatever they might be in the eyes of policymakers, clearly have won over the supporters of a federal response to climate change.

Price instruments appear to have potential only in a few areas. The most likely adoption of a pure price approach is for traffic congestion. Congestion pricing and tolls seem far more likely to be used to reduce congestion delays than any kind of quantity instrument.

A price-type approach is also being used to increase use of renewable electricity, through Renewable Portfolio Standards. It is notable that in this case, the regulation and accompanying discussion have been framed in terms of a cap-and-trade approach. Only economists will recognize that this is essentially a form of taxes.

2. Hybrid policies are common. A hybrid policy is a cap-and-trade system that contains a safety value provision, in which extra allowances are made available at a pre-announced price. Safety-valves appear in some form in the Acid Rain Trading Program, the Clear Skies proposal, CAFE, renewable portfolio standards, and many water quality trading systems. A safety-valve price was considered for the Clean Air Mercury Rule but abandoned in the final rule. Current U.S. policies exhibit a range of safety-valves, from those that are invoked routinely to those that would be invoked only in rare circumstances.

3. Market mechanisms are almost never the sole instrument used to regulate environmental behavior. They are almost always combined with other regulations, in stark contrast to the theoretical literature which, with equal uniformity, models market mechanisms as the sole regulation that polluters face. Thus, much of the theory of real-world “mixed” approaches remains undeveloped.

An important corollary of our conclusion is that non-market approaches remain pervasive throughout U.S. environmental policy.

Non-market elements show up *even in the most market-oriented of policies*, such as the Acid Rain Trading Program's subsidy for early adoption of scrubbers and its continued requirement of the inefficient New Source Review and New Source Performance Standards. Likewise, the ITQ approach to fishery management has not been uniformly accompanied by a removal of inefficient gear or vessel capacity restrictions. Other market stalwarts, such as the Lead Trading Program and the Ozone-Depleting Chemicals program, simultaneously contained numerous restrictions on trades and on products.

4. Cap-and-trade often does not cap a true externality or cap the externality it appears to be aimed at. Mileage standards for new cars do not cap gasoline consumption, which appears to be the law's target. Renewable Portfolio Standards promote renewable electricity but do not explicitly penalize the air quality or carbon content of nonrenewable electricity. Under tradable development rights, it is the underlying zoning regulation that effectively caps the number of farmland acres that may be converted to development but this cap is never binding even though the price of a development right is positive.

Note that many of the arguments for the transparency of market mechanisms are no longer valid when the "wrong" item is capped. Mileage standards probably reduce gasoline consumption, but their effects on such consumption are difficult to tease out because gasoline consumption is not capped and the connection between new car mileage and aggregate gasoline consumption (by both new and old cars) is complex. If the cap-and-trade applied directly to gasoline, the effects would be easier to identify and predict.

The misplaced-cap issue arises, at times, because legislators or regulators are attempting to remedy several different externalities simultaneously. Therefore, policy design in the presence of multiple externalities deserves much more research attention. Optimal regulation of multiple externalities typically requires multiple instruments. If only a single instrument can be used, it is not clear what regulatory instrument is best, although we presume that it will involve a market-based approach.

5. Non-point source trading for water quality remains one of the most active areas for new markets, but thin markets, the inability to

monitor discharges, and the fact that non-point sources are unregulated create a situation that is far from the market paradigm envisioned by Montgomery (1972) or Baumol and Oates (1988). The latter two issues also arise for carbon sequestration.

There are two broad issues to consider. First, although the markets are inherently thin, there may be some ways to make them operate more efficiently. Regulators could adopt some sort of hybrid approach, as used in the Long Island Sound or Tar-Pamlico programs. Under this approach, the safety valve, which regulators and sources may then *expect* to be invoked, eliminates the thin-market problems of market power, high transactions costs, and price uncertainty. Alternatively, regulators could identify concrete, short-term activities, such as the planting of a winter cover crop, that non-point sources could engage in for credits. The simplicity and observability of this action would allow more non-point sources to participate and with much lower transactions costs.

Second, because non-point sources are largely unregulated, their participation in a market invokes the *opt-in* problem. Regulators must determine a baseline level of discharges. This baseline is difficult to determine, yet if it is calculated incorrectly – either too high or too low – the regulation will be inefficient, possibly dramatically so. This problem arises in many circumstances beyond water pollution, and especially so in the context of allowing developing countries to opt in to a global agreement on greenhouse gas limits, rather than requiring specific reductions from them. The problem also arises for greenhouse gas regulation within the U.S., since landowners who undertake reforestation or other carbon-sequestration actions may want to opt-in to a domestic carbon cap-and-trade program. Understanding and solving the opt-in problem remain key challenges for environmental economists.

5.2 Final remarks

Economists have written extensively and for many decades on the advantages of market-based approaches to environmental problems. As a result of this long-standing support – and, presumably, the correctness of most of the economists' claims – policies that contain at least

some elements of a market approach have now been applied to many environmental concerns.

Further improvement in market mechanisms is likely to occupy the attention of economists and policymakers in the coming decade. Non-market elements remain part of even the most market-oriented of policies. Economists need to examine the implications of and justifications for such elements and, where needed, press for their elimination.

Many cap-and-trade systems also do not focus on the item that they appear to be aimed at. The economics of this “misplaced-cap” problem are worth exploring further. In some cases the disjunction occurs because the true externality-causing item cannot be feasibly monitored. Future improvements in monitoring capabilities are likely and should greatly enhance the government’s ability to set-up efficient market instruments aimed at the true externality. In the cases where a cap-and-trade could at present be imposed on the true externality, economists should push for such a shift.

Finally, many remaining environmental problems arise from non-point sources, which do not initially appear to be amenable to market-based approaches. The U.S. experience with burgeoning point-non-point trading for water quality should help economists and policymakers learn how market approaches might be used even for non-point sources. The economics of the opt-in problem that occurs in these situations also remains under-explored.

Although there is a rich literature on the features and applicability of market approaches to regulating the environment, we believe the growing roles of these regulations present new, real-world issues that warrant further analysis. Our scrutiny of the existing and proposed environmental regulations demonstrates that there are still many areas in which economists can help improve environmental policies.

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