## Chapter 4

# Regulatory and Non-Regulatory Approaches to Pollution Control

This chapter briefly describes several regulatory and non-regulatory approaches used in environmental policy making. The goals of this chapter are to introduce several important analytic terms, concepts, and approaches; to describe the conceptual foundations of each approach; and to provide additional references for those interested in a more in-depth discussion.[1] Specifically, this chapter discusses the following four general approaches to environmental policy making: (1) command-and-control regulation; (2) market-based incentives; (3) hybrid approaches; and (4) voluntary initiatives. While command-and-control regulations have been a commonly used method of environmental regulation in the United States, EPA also employs the three other approaches. Market-based incentives and hybrid approaches offer the regulated community an opportunity to meet standards with increased flexibility and lower costs compared to many command-and-control regulations, while voluntary initiatives may allow environmental improvements in areas not traditionally regulated by EPA. The chapter also includes a discussion of criteria used to evaluate the effectiveness of regulatory and non-regulatory approaches to pollution control.

## 4.1 Evaluating Environmental Policy

Once federal action is deemed necessary to address an environmental problem, policy makers have a number of options at their disposal to influence pollution levels. In deciding which approach to implement, policy makers must be cognizant of constraints and limitations of each approach in addressing specific environmental problems. It is important to account for how political and information constraints, imperfect competition, or pre-existing market distortions interact with various policy options. Even when a particular approach is appealing from a social welfare perspective, it may not be consistent with statutory requirements, or may generate additional concerns when considered along with other existing regulations. While any policy option under consideration must balance cost considerations with other important policy goals (including benefits), economic efficiency and cost-effectiveness are two economic concepts useful for framing the discussion and comparison of the regulatory options presented in the remaining sections of this chapter.

### 4.1.1 Economic Efficiency

Economic efficiency can be defined as the maximization of social welfare. An efficient market is one that allows society to maximize the net present value (NPV) of benefits: the difference between a stream of social benefits and social costs over time. The efficient level of production is referred to as *Pareto optimal* because there is no way to rearrange production or reallocate goods in such a way that someone is better off without making someone else worse off in the process. The efficient

1 Baumol and Oates (1988), particularly Chapters 10-14; Kahn (1998); Kolstad (2000); Sterner (2003); and Field and Field (2005) are useful references on the economic foundations of many of the approaches presented here.

level of production occurs without government intervention in a market characterized by no market failures or externalities (see Appendix A for a more detailed discussion of efficiency and for a graphical representation of the efficient point of production). Government intervention may be justified, however, when a market failure or externality exists (see Appendix A), in which case the government may attempt to determine the socially optimal point of production once such externalities have been internalized. Said differently, government analysts may evaluate which of the various policy approaches under consideration maximizes the benefits of reducing environmental damages, net the resulting abatement costs.

Conceptually, the socially optimal level is determined by reducing emissions until the benefit of abating one more unit of pollution (i.e., the marginal abatement benefit) — measured as a reduction in damages — is equal to the cost of abating one additional unit (i.e., the marginal abatement cost).[2] In the simplest case, when each polluter chooses the level at which to emit according to this decision rule (i.e., produce at a level at which the marginal abatement benefit is equal to the marginal abatement cost), an efficient aggregate level of emissions is achieved when the cost of abating one more unit of pollution is equal across all polluters. Any other level of emissions would result in a reduction in net benefits.

This definition of efficiency describes the simplest possible world where a pollutant is a uniformly mixed flow pollutant — the pollutant does not accumulate or vary over time — and the marginal damages that result are independent of location. When pollution levels and damages vary by location, the efficient level of pollution is achieved when marginal abatement costs adjusted by individual transfer coefficients are equal across all polluters. Temporal variability also implies an adjustment to this equilibrium condition. In the case of a stock pollutant, marginal abatement costs are equal across the discounted sum of damages from today's emissions in all future time periods. In the case of a flow pollutant, this condition should be adjusted to reflect seasonal or daily variations. Under uncertainty, it is useful to think of the efficient level of pollution as a distribution instead of as a single point estimate.

The reality of environmental decision making is that Agency analysts are rarely in the position to select the economically efficient point of production when designing policy. This is partly because the level of abatement required to reduce a particular environmental problem is often determined legislatively, while the implementation of the policy to achieve such a goal is left to the Agency. In cases where the Agency has some say in the stringency of a policy, its degree of flexibility in determining the approach taken varies by statute. This may limit its ability to consider particular approaches or to use particular policy instruments. It is also important to keep in mind analytic constraints. In cases where it is particularly difficult to quantify benefits, cost-effectiveness may be the most defensible analytic framework.

### 4.1.2 Cost-Effectiveness

The efficiency of a policy option differs from its cost-effectiveness. A policy is cost-effective if it meets a given goal at least cost, but cost-effectiveness does not encompass an evaluation of whether that goal has been set appropriately to maximize social welfare. All efficient policies are cost-effective, but it is not necessarily true that all cost-effective policies are efficient. A policy is considered cost-effective when marginal abatement costs are equal across all polluters. In other words, for any level of total abatement, each polluter has the same cost for their last unit abated.

## 4.2 Traditional Command-and-Control or Prescriptive Regulation

Many environmental regulations in the United States are prescriptive in nature (and are often

2 The idea that a given level of abatement is efficient — as opposed to abating until pollution is equal to zero — is based on the economic concept of diminishing returns. For each additional unit of abatement, marginal social benefits decrease while marginal social costs of that abatement increase. Thus, it only makes sense to continue to increase abatement until the point where marginal benefits and marginal costs are just equal. Any abatement beyond that point will incur more additional costs than benefits.

referred to as command-and-control regulations).[3] A prescriptive regulation can be defined as a policy that prescribes how much pollution an individual source or plant is allowed to emit and/or what types of control equipment it must use to meet such requirements. Such a standard is often defined in terms of a source-level emissions rate. Despite the introduction of potentially more cost-effective methods for regulating emissions, this type of regulation is still commonly used and is sometimes statutorily required. It is almost always available as a "backstop" if other approaches do not achieve desired pollution limits.

Because a prescriptive standard is commonly defined in terms of an emissions *rate*, it does not directly control the aggregate emission *level*. In such cases, aggregate emissions will depend on the number of polluters and the output of each polluter. As either production or market size increase, so will aggregate emissions. Even when the standard is defined in terms of an emission level per polluting source, aggregate emissions will still be a function of the total number of polluters.

When abatement costs are similar across regulated sources, a source-level standard may be reasonably cost-effective. However when abatement costs vary substantially across polluters, reallocating abatement activities so that some polluters have stricter standards than others could lead to substantial cost savings. If reallocation were possible (e.g., through a non-prescriptive approach), a polluter facing relatively high abatement costs would continue to emit at its current level but would pay for the damages incurred (e.g., by paying a tax or purchasing permits), while a polluter with relatively low abatement costs would reduce its emissions.

Note that regulators can at least partially account for some variability in costs by allowing prescriptive standards to vary according to size of the polluting entity, production processes, geographic location, or other factors. Beyond this, however, a prescriptive standard usually does not allow for reallocation of abatement activities to take place — each entity is still expected to achieve a specified emissions standard. Thus, while pollution may be reduced to the desired level, it is often accomplished at a higher cost under a prescriptive approach.[4]

It is common to "grandfather," or exempt, older polluters from new prescriptive regulations, thereby subjecting them to a less stringent standard than newer polluters. Grandfathering creates a bias against constructing new facilities and investing in new pollution control technology or production processes.[5] As a result, grandfathered older facilities with higher emission rates tend to remain active longer than they would if the same emissions standard applied to all polluters.

The most stringent form of prescriptive regulation is one in which the standard specifies zero allowable source-level emissions. For instance, EPA has completely banned or phased out the use or production of chlorofluorocarbons (CFCs) and certain pesticides. This approach to regulation is potentially useful in cases where the level of pollution that maximizes social welfare is at or near zero.[6]

Two types of prescriptive regulations exist: technology or design standards; and performance-based standards.

### 4.2.1 Technology or Design Standards

A *technology* or *design standard*, mandates the specific control technologies or production

3 Goulder and Parry (2008) refer to these as "direct regulatory instruments" because they feel that "command-and-control" has a "somewhat negative connotation." Ellerman (2003) refers to them as prescriptive regulations. We follow that convention here. Notable exceptions to this type of regulation in the U.S. experience include the phase-down in lead content in gasoline, which allowed trading of credits among refineries and offset programs applied in non-attainment areas. For more information on early applications of market incentives, see U.S. EPA (2001b).

4 See Tietenberg (2004) for a discussion of empirical studies that examine the cost-effectiveness of prescriptive air pollution regulations. Of the ten studies included, eight found that prescriptive regulations cost at least 78 percent more than the most cost-effective strategy.

5 For a discussion of grandfathering, see Helfand (1991).

6 For cases where the optimal level of pollution is at or near zero, the literature also indicates that market-based incentives can sometimes be useful as a transition instrument for the phasing-out of a particular chemical or pollutant. See Sterner (2003) and Kahn (1998).

### Text Box 4.1 - Coase Solution

Government intervention for the control of environmental externalities is only necessary when parties cannot work out an agreement between themselves. Coase (1960) outlined conditions under which a private agreement between affected parties might result in the attainment of a social welfare maximizing level of pollution without government intervention. First, property rights must be clearly defined. In situations where the resource in question is not “owned” by anyone, there are no incentives to negotiate, and the offending party can “free ride,” or continue to pollute, without facing the costs of its behavior.

When property rights have been allocated, a social welfare maximizing solution can be reached regardless of which party is assigned the property rights, although the equity of the assignment may vary. Take for example a farm whose pesticide application to its crops contributes pollution to the well water of nearby homeowners. If property rights of the watershed are assigned to the homeowners, then the farm may negotiate with the homeowners to allow it to continue to use the pesticide. The payment need not be in the form of cash but could be payments in kind. If property rights of the watershed are given to the farm, then the homeowners would have to pay the farm to stop applying the pesticide.

In each case, the effectiveness of the agreement is contingent on meeting additional conditions: bargaining must be possible, and transaction costs must be low. These conditions are more likely to be met when there are only a small number of individuals involved. If either party is unwilling to negotiate or faces high transaction costs, then no private agreement will be reached. Asymmetric information can also hinder the ability of one or more party to come to an agreement. Going back to the example, consider a case where there are many farms in the watershed using the pesticide on their crops. Clearly homeowners would have more difficulty in negotiating an agreement with every farm than they would when negotiating with one farm.

processes that an individual pollution source must use to meet the emissions standard. This type of standard constrains plant behavior by mandating how a source must meet the standard, regardless of whether such an action is cost-effective. Technology standards may be particularly useful in cases where the costs of emissions monitoring are high but determining whether a particular technology or production process has been put in place to meet a standard is relatively easy. However, since these types of standards specify the abatement technology required to reduce emissions, sources do not have an incentive to invest in more cost-effective methods of abatement or to explore new and innovative abatement strategies or production processes that are not permitted by regulation.

## 4.2.2 Performance-based Standards

A *performance-based standard* also requires that polluters meet a source-level emissions standard, but allows a polluter to choose among available methods to comply with the standard. At times, the available methods are constrained by additional criteria specified in a regulation. Performance-based standards that are technology-based do not specify a particular technology, but rather consider what is possible for available and affordable technology to achieve when establishing a limit on emissions.[7]

In the case of a performance-based standard, the level of flexibility a source has in meeting the standard depends on whether the standard specifies an emission *level* or an emission *rate* (i.e., emissions per unit of output or input). A standard that specifies an emission level allows a source to

7 As an example, Reasonably Available Control Technology (RACT) specifies that the technology used to meet the standard should achieve “the lowest emission limit that a particular source or source category is capable of meeting by application of control technology that is reasonably available considering technological and economic feasibility.” RACT defines the standard on a case-by-case basis, taking into account a variety of facility-specific costs and impacts on air quality. EPA has been restrictive in its definition of technologies meeting this requirement and eliminates those that are not commercially available (see Swift 2000).

choose to implement an appropriate technology, change its input mix, or reduce output to meet the standard. An emission rate, on the other hand, may be more restrictive depending on how it is defined. If the emissions rate is defined per unit of output, then it does not allow a source to meet the standard through a reduction in output. If the standard is defined as an average emissions rate over a number of days, then the source may still reduce output to meet the standard.

The flexibility of performance-based standards encourages firms to innovate to the extent that they allow firms to explore cheaper ways to meet the standard; however, they generally do not provide incentives for firms to reduce pollution beyond what is required to reach compliance.[8] For emissions that fall below the amount allowed under the standard, the firm faces a zero marginal abatement cost since the firm is already in compliance. Also, because permitting authority is often delegated to the States, approval of a technology in one state does not ensure its use is allowed in another. Firm investment in research to develop new, less expensive, and potentially superior technologies is therefore discouraged.[9]

## 4.3 Market-Oriented Approaches

**Market-oriented approaches** (or market-based approaches) create an incentive for the private sector to incorporate pollution abatement into production or consumption decisions and to innovate in such a way as to continually search for the least costly method of abatement.[10] Market-oriented approaches can differ from more traditional regulatory methods in terms of economic efficiency (or cost-effectiveness) and the distribution of benefits and costs. In particular, many market-based approaches minimize polluters' abatement costs, an objective that often is not achieved under command-and-control based approaches. Because market-based approaches do not mandate that each polluter meet a given emissions standard, they typically allow firms more flexibility than more traditional regulations and capitalize on the heterogeneity of abatement costs across polluters to reduce aggregate pollution efficiently. Environmental economists generally favor market-based policies because they tend to be least costly, they place lower information burden on the regulator, and they provide incentives for technological advances. Four classic market-based approaches are discussed in this section:

- Marketable permit systems;
- Emission taxes;
- Environmental subsidies; and
- Tax-subsidy combinations. [11]

While operationally different (e.g., taxes and subsidies are price-based while marketable permits are quantity-based), these market-based instruments are more or less functionally equivalent in terms of the incentives they put in place. This is particularly true of emission taxes and cap-and-trade systems, which can be designed to achieve the same goal at equivalent cost. The sections that follow discuss each of these market-based approaches in turn.

### 4.3.1 Marketable Permit Systems

Several forms of emissions trading exist, including cap-and-trade systems, project-based trading

8 For a theoretical analysis of incentives for technological change, see Jung et al. (1996) and Montero (2002). Empirical analyses can be found in Jaffe and Stavins (1995), and Kerr and Newell (2003).

9 See Swift (2000) and U.S. EPA (1991) for a detailed discussion of how emission rate-based standards hinder technological innovation.

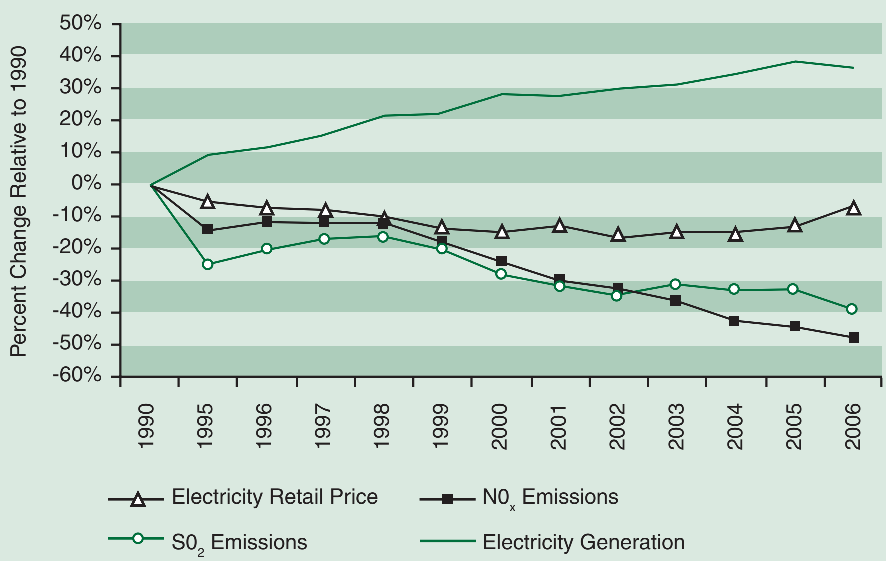
10 The incentive to innovate means that the marginal abatement cost curve shifts downward over time as cheaper abatement options are introduced.

11 The literature on applied market-based approaches for environmental protection should be consulted, along with the references they contain, for information concerning the design, operation, and performance of these approaches. Anderson and Lohof (1997) and Stavins (1998a, 2000b) compile information on both the theory and empirical use of economic incentives. Newell and Stavins (2003) generate rules-of-thumb designed to make it easy for policy makers to determine when market-based incentives may result in cost savings over command-and-control regulations. Harrington et al. (2004) compare the costs and outcomes of command-and-control and incentives-based regulatory approaches to the same environmental problem in the United States and Europe. Additional sources include Sterner (2003), Stavins (2003), Tietenberg (1999, 2002), U.S. EPA (2004a, 2001a), OECD (1994a, 1994b), and proceedings published under the "Project 88" forum, Stavins (1988, 1991).

## Text Box 4.2 - Acid Rain Trading Program for Sulfur Dioxide ($SO_2$)

In 1995, Title IV of the 1990 Clean Air Act Amendments established a cap-and-trade system for $SO_2$ emissions to address the problem of acid rain. Two hundred and sixty three of the highest emitting $SO_2$ units of 110 electricity-generating plants were selected to participate in the first phase of the trading program. Emissions of $SO_2$ in 1995 were initially limited to 8.7 million tons for those facilities. Of the plants that participated, most were coal-fired units located east of the Mississippi River. Under this system, allowances were allocated to units on a historical basis, after which they could use the allowances, sell them to other units, or “bank” the allowances for use in subsequent years. Continual emission monitoring (CEM) systems have allowed the government to easily monitor and enforce emission restrictions in accordance with the allowances. The second phase of the program, initiated in 2000, imposed a national $SO_2$ emissions cap of 10 million tons and brought almost all $SO_2$ generating units into the system.

Initial evaluations of the first phase of implementation suggest that the $SO_2$ trading system has significantly reduced emissions at a relatively low cost. In fact, allowance prices have been considerably lower than predicted, reflecting lower than expected marginal costs. A significant level of trading has occurred and has resulted in savings of over $1 billion per year as compared to command-and-control alternatives. Emissions in 1995 were almost 40 percent below the 10 million ton limit. The evaluations demonstrated that one reason for such large reductions in $SO_2$ emissions below the allowable limit is the ability to bank allowances for future use. The success of the program has continued into the second phase, with recent estimates of the full U.S. Acid Rain Program’s benefits [including $SO_2$ trading and direct nitrogen oxide ($NO_x$) controls] reaching upwards of $120 billion annually in 2010 with annual costs around $3 billion (in 2000$); a benefit to cost ratio of about 40 to 1. Trends over the life of the program show that while electricity generation has grown steadily and $SO_2$ and $NO_x$ emissions have fallen substantially, electricity retail prices, until very recently, have declined in real terms.




*Source: U.S. EPA 2007a*

**For more information, see Burtraw and Bohi (1997), Schmalensee et al. (1998), Stavins (1998b, 2003), Carlson et al. (2000), Chestnut and Mills (2005), and U.S. EPA (2007a).**

systems and emissions rate trading systems. The common element across these programs is that sources are able to trade credits or allowances so that those with opportunities to reduce emissions at lower costs have an incentive to do so. Each of these systems is discussed in turn below.[12]

12 For a more detailed discussion of the various systems and how to design them, see U.S. EPA (2003c).

#### 4.3.1.1 Cap-and-Trade Systems

In a cap-and-trade system the government sets the level of aggregate emissions, emission allowances are distributed to polluters, and a market is established in which allowances may be bought or sold. The price of emission allowances is allowed to vary. Because different polluters incur different private abatement costs to control emissions, they are willing to pay different amounts for allowances. Therefore, a cap-and-trade system allows polluters who face high marginal abatement costs to purchase allowances from polluters with low marginal abatement costs, instead of installing expensive pollution control equipment or using more costly inputs. Cap-and-trade systems also differ from command-and-control regulations in that they aim to limit the aggregate emission level over a compliance period rather than establish an emissions rate.

If the cap is set appropriately, then the equilibrium price of allowances, in theory, adjusts so that it equals the marginal external damages from a unit of pollution. This equivalency implies that any externality associated with emissions is completely internalized by the firm. For polluters with marginal abatement costs greater than the allowance price, the cheapest option is to purchase additional units and continue to emit. For polluters with marginal abatement costs less than the allowance price, the cheapest option is to reduce emissions and sell their permits. As long as the price of allowances differs from individual firms' marginal abatement costs, firms will continue to buy or sell them. Trading will occur until marginal abatement costs equalize across all firms.[13]

Generally, allowances initially sold at auction represent income transfers from the purchasers to the government in the amount of the price paid for the allowances. The collection of revenue through this method of allowance allocation gives the government the opportunity to reduce pre-existing market inefficiencies, to reduce distributional consequences of the policy, or to invest in other social priorities. Allowances may also be allocated to polluters according to a specified rule. This represents a transfer from the government to polluting firms, some of which may find that the value of allowances received exceeds the firm's aggregate abatement costs.

The distribution of rents under cap-and-trade systems should be considered when comparing these systems with more traditional regulatory approaches. If the allowances are auctioned or otherwise sold to polluters, the distributional consequences will be similar to those experienced when regulating using taxes. If allowances are distributed for free to polluters, however, distributional consequences will depend on the allocation mechanism (e.g., historical output or inputs), on who receives the allowances, and on the ability of the recipients to pass their opportunity costs on to their customers. If new entrants must obtain allowances from existing polluters, then the policy maker should also consider potential barrier-to-entry effects. Differing treatment applied to new versus existing polluters can affect the eventual distribution of revenues, expenses, and rents within the economy.

Additional considerations in designing an effective cap-and-trade system include "thin" markets, transaction costs, banking, effective monitoring, and predictable consequences for noncompliance. The United States' experience suggests that a market characterized by low transaction costs and being "thick" with buyers and sellers is critical if pollution is to be reduced at the lowest cost. This is because small numbers of potential traders in a market make competitive behavior unlikely, and fewer trading opportunities result in lower cost savings. Likewise, the number of trades that occur could be significantly hindered by burdensome requirements that increase the transaction costs associated with each trade.[14]

13 The U.S. Acid Rain Program established under Title IV of the 1990 Clean Air Act Amendments is a good example of a marketable permit program. For economic analyses of this program see Joskow et al. (1998), Stavins (1998b), Ellerman et al. (2000), and Chestnut and Mills (2005). For more information on the program itself see Text box 4.2 and EPA's (2008a) Acid Rain Website at http://www.epa.gov/acidrain (accessed April 5, 2004).

14 This is also often the case for bubbles and offsets. See O'Neil (1983) for an evaluation of an early example of a permit-trading program in the United States and the main reasons for its failure.

Cap-and-trade systems should also be sensitive to concerns about potential temporal or spatial spikes (i.e., hotspots — areas in which the pollution level has the potential to increase as a result of allowance trading). This may happen, for example, in an area in which two facilities emit the same amount of pollution, but due to differences in exact location and site characteristics, one facility's impact on environmental quality differs substantially from that of the other polluter. While one potential solution to this problem is to adjust trading ratios to equalize the impact of particular polluters on overall environmental quality, determining the appropriate adjustments to these ratios can be costly and difficult. Other possible solutions include zone-based trading and establishing pollution "floors."

Two recent reviews of the literature (Burtraw et al. 2005 and Harrington et al. 2004) find little evidence of spatial or temporal spikes in pollution resulting from the use of market-based approaches. In fact, market-based approaches have led to smoothing of emissions across space in some cases. These results come primarily from studies of the $SO_2$ and $NO_x$ trading programs and if the market-based policy is not carefully designed, the results may not transfer to other pollutants that have more localized effects.

Banking introduces increased flexibility into a trading system by allowing polluters to bank unused permits for future use. A firm may reduce emissions below the allowance level now, and bank (or save) remaining allowances to cover excess emissions or sell to another polluter at a later time. In this way, polluters that face greater uncertainty regarding future emissions, or that expect increased regulatory stringency, can bank allowances to offset potentially higher future marginal abatement costs.

For a cap-and-trade system to be effective, reliable measurement and monitoring of emissions must occur with predictable consequences for noncompliance. At the end of the compliance period, emissions at each source are compared to the allowances held by that source. If a source is found to have fewer allowances than the monitored emission levels, it is in noncompliance and the source must provide allowances to cover its environmental obligation. In addition, the source must pay a penalty automatically levied per each ton of excess emissions.[15]

#### 4.3.1.2 Project-Based Trading Systems

Offsets and bubbles (sometimes known as "project-based" trading systems) allow restricted forms of emissions trading across or within sources to allow sources greater flexibility in complying with command-and-control regulations such as emission limits or facility-level permits. An offset allows a new polluter to negotiate with an existing source to secure a reduction in the latter's emissions. A bubble allows a facility to consider all sources of emissions of a particular pollutant within the facility to achieve an overall target level of emissions or environmental improvement. While offsets and bubbles are mostly used to control air pollution in non-attainment areas, they have been historically hindered by high administrative and transaction costs because they require case-by-case negotiation to convert a technology or emission rate limit into tradable emissions per unit of time, to establish a baseline, and to determine the amount of credits generated or required (U.S. EPA 2001a).

#### 4.3.1.3 Rate-Based Trading Systems

Rather than establish an emissions cap, the regulatory authority under a rate-based trading program, establishes a performance standard or emissions rate. Sources with emission rates below the performance standard can earn credits and sell them to sources with emission rates above the standard. As with the other trading systems, sources able to improve their emissions rate at low cost have an incentive to do so since they can sell the resulting credits to those sources facing higher costs of abatement. However, emissions may increase under these programs if sources increase their utilization or if new sources enter the market. Therefore, the regulating authority

15 Notably, the U.S. Acid Rain Trading Program has nearly 100 percent compliance and requires only about 50 EPA staff to administer.

may need to periodically impose new rate standards to achieve and maintain the desired emission target, which in turn may lead to uncertainty in the long term for the regulated sources. Rate-based trading programs have been used in the United States to phase out lead in gasoline (1985) and to control mobile source emissions (U.S. EPA 2003c).

### 4.3.2 Emissions Tax

**Emissions taxes** are exacted per unit of pollution emitted and induce a polluter to take into account the external cost of its emissions. Under an emissions tax, the polluter will abate emissions up to the point where the additional cost of abating one more unit of pollution is equal to the tax, and the tax will result in an efficient outcome if it is set equal to the additional external damage caused by the last unit of pollution emitted.

As an example of how an emissions tax works, suppose that emissions of a toxic substance are subject to an environmental charge based on the damages the emissions cause. To avoid the emissions tax, polluters find the cheapest way to reduce pollution. This may involve a reduction in output, a change in inputs to production, the installation of pollution control equipment, or a process change that prevents the creation of pollution. Polluters decide individually how much to control their emissions, based on the costs of control and the magnitude of the tax. The polluting firm reduces emissions to the point where the cost of reducing one more unit of emissions is just equal to the tax per unit of emissions. For any remaining emissions, the polluter prefers to pay the tax rather than to abate further. In addition, the government earns revenue that it may use to reduce other pollution or reduce other taxes, or may redistribute to finance other public services.[16] While difficult to implement in cases where there is temporal and/or spatial variation in emissions, policy makers can more closely approximate the ambient impact of emissions by incorporating adjustment factors for seasonal or daily fluctuations or individual transfer coefficients in the tax.

Despite the apparent usefulness of such a tax, true emissions taxes — those set equal or close to marginal external damages — are relatively rare in the United States.[17] This is because taxing emissions directly may not be feasible when emissions are difficult to measure or accurately estimate, when it is difficult to define and monetarily value marginal damages from a unit of emissions (which is needed to properly set the tax), or when taxes are applied to emissions that are difficult to monitor and/or enforce. In addition, attempts to measure and tax emissions may lead to illegal dumping.[18] Other considerations when contemplating the use of emission taxes include the potential imposition of substantially different cost burdens on polluters as compared with other regulatory approaches, political incentives to set the tax too low, and the collection of revenues and distribution of economic rents that result from such programs.

User or product charges are a variation on emission taxes that are occasionally utilized in the United States. These charges may be imposed directly upon users of publicly operated facilities or upon intermediate or final products whose use or disposal harms the environment. User or product charges may be effective approximations of an emissions tax for those cases in which the product taxed is closely related to emissions. User charges have been imposed on firms that discharge waste to municipal wastewater treatment facilities and on non-hazardous solid wastes disposed of in publicly-operated landfills. Product charges have been imposed on products that release CFCs into the atmosphere, that utilize more gasoline (such as cars), or require more fertilizer. In practice, both user and product charges are usually set at a level only sufficient to recover the *private costs* of operating the public system, rather than being set at a level selected to create proper incentives for reducing pollution to the socially optimal level.

16 For more information on how the government can use revenues from taxes to offset distortions created by other taxes, see Goulder (1995) and Goulder et al. (1997).

17 These taxes are called "Pigovian" after the economist, Arthur Pigou, who first formalized them. See Pigou (1932).

18 See Fullerton (1996) for a discussion of the advantages and disadvantages of emission taxes.

Taxes and charges facilitate environmental improvements similar to those that result from marketable permit systems. Rather than specifying the total quantity of emissions, however, taxes, fees, and charges specify the effective "price" of emitting pollutants.

### 4.3.3 Environmental Subsidies

**Subsidies** paid by the government to firms or consumers for per unit reductions in pollution create the same abatement incentives as emission taxes or charges. If the government subsidizes the use of a cleaner fuel or the purchase of a particular control technology, firms will switch from the dirtier fuel or install the control technology to reduce emissions up to the point where the private costs of control are equal to the subsidy. It is important to keep in mind that an environmental subsidy is designed to correct for an externality not already taken into account by firms when making production decisions. This type of subsidy is fundamentally different from the many subsidies already in existence in industries such as oil and gas, forestry, and agriculture, which exist for other reasons apart from environmental quality, and therefore can exacerbate existing environmental externalities.

Unlike an emissions tax, a subsidy lowers a firm's total and average costs of production, encouraging both the continued operation of existing polluters that would otherwise exit the market, and the entry into the market by new firms that would otherwise face a barrier to entry. Given the potential entrance of new firms under a subsidy, the net result may be a decrease in pollution emissions from individual polluters but an increase in the overall amount.[19] For this reason, subsidies and taxes may not have the same aggregate social costs, or result in the same degree of pollution control. A subsidy also differs from a tax because it requires government expenditure. Analysts should always consider the opportunity costs associated with using public funds.

It is possible to minimize the entry and exit of firms resulting from subsidies by redefining the subsidy as a partial repayment of verified abatement costs, instead of defining it as a per unit payment for emissions reductions relative to a baseline. Under this definition, the subsidy now only relates to abatement costs incurred and does not shift the total or average cost curves, thereby leaving the entry and exit decisions of firms unaffected. Defining the subsidy in this way also minimizes strategic behavior because no baseline must be specified. [20]

Instead of pursuing a per unit emissions subsidy, the government may choose to lower the private costs of particular actions to the firm or consumer through cost sharing. For example, if the government wishes to encourage investment in particular pollution control technologies, the subsidy may take the form of reduced interest rates, accelerated depreciation, direct capital grants, and loan assistance or guarantees for investments. Cost-sharing policies alone may not induce broader changes in private behavior. In particular, such subsidies may encourage investment in pollution control equipment, rather than encouraging other changes in operating practices such as recycling and reuse, which may not require such costly capital investments. However, in conjunction with direct controls, pollution taxes, or other regulatory mechanisms, cost sharing may influence the nature of private responses and the distribution of the cost burden. As is the case with emissions taxes, subsidy rates also can be adjusted to account for both spatial and temporal variability.

A government "buy-back" constitutes another type of subsidy. Under this system, the government either directly pays a fee for the return of a product or subsidizes firms that purchase recycled materials. For instance, consumers may be offered

19 See Sterner (2003) for a more in-depth discussion of how subsidies work and for numerous examples of subsidy programs in the United States and other countries.

20 Strategic behavior is a problem common to any instrument or regulation that measures emissions relative to a baseline. In cases where a firm or consumer may potentially receive funds from the government, they may attempt to make the current state look worse than it actually is, in order to receive credit for large improvements. If firms or consumers are responsible for paying for certain emissions above a given level, they may try to influence the establishment of that level upward in order to pay less in fines or taxes.

a cash rebate on the purchase of a new electric or push mower when they scrap their old one. The rebate is earned when the old gasoline mower is turned in and a sales receipt for the new device is provided.[21] Buy-back programs also exist to promote the scrapping of old, high-emission vehicles.

Environmental subsidies in the United States have been used to encourage proper waste management and recycling by local governments and businesses; the use of alternative fuel vehicles by public bus companies, consumers, and businesses; and land conservation by property owners using cost-sharing measures. While most of these subsidies are not defined per unit of emissions abated, they can be effective when the behavioral changes they encourage are closely related to the use of products with reduced emissions.

### 4.3.4 Tax-Subsidy Combinations

Emission taxes and environmental subsidies can also be combined to achieve the same level of abatement as achieved when the tax and subsidy instruments are used separately. One example of this type of instrument is referred to as a **deposit-refund system** in which the deposit operates as a tax and the refund serves as a partially offsetting subsidy. As with the other market instruments already discussed, a deposit-refund system creates economic incentives to return a product for reuse or proper disposal, or to use a particular input in production, provided that the deposit exceeds the private cost of returning the product or switching inputs.

Under the deposit-refund system, the deposit is applied to either output or consumption, under the presumption that all production processes of the firm pollute or that all consumption goods become waste. A refund is then provided to the extent that the firm or consumer provides proof of the use of a cleaner form of production or of proper disposal. In the case where a deposit-refund is used to encourage firms to use a cleaner input, the deposit on output induces the firm to reduce its use of *all* inputs, both clean and dirty. The refund, however, provides the firm with an incentive to switch a specific input or set of inputs that result in a refund, such as a cleaner fuel or a particular pollution control technology.

A tax and offsetting subsidy combination functions best when it is possible to discern a direct relationship between an input, or output, and emissions. For instance, a tax on the production or use of hydrochlorofluorocarbons (HCFCs) combined with a refund for HCFC recycled or collected in a closed system is a good proxy for a direct emissions tax on ozone depletion.[22]

The most common type of tax-subsidy combination is the deposit-refund system, which is generally designed to encourage consumers to reduce litter and increase the recycling of certain components of municipal solid waste.[23] The most prominent examples are deposit-refunds for items such as plastic and glass bottles, lead acid batteries, toner cartridges and motor oil. Other countries have implemented deposit-refund systems on a wider range of products and behaviors that contribute to pollution, including the sulfur content of fuels (Sweden), product packaging (Germany), and deforestation (Indonesia). Tax-subsidy combinations have also been discussed in the literature as a means of controlling nonpoint source water pollution, cadmium, mercury, and the removal of carbon from the atmosphere.[24]

The main advantage of a combined tax and subsidy is that both parts apply to a market transaction. Because the taxed and subsidized items are easily observable in the market, this type of economic instrument may be particularly appealing when it is difficult to measure emissions or to control illegal dumping. In addition, polluters have an incentive to reveal accurate information on abatement activity to qualify for the subsidy.

21 For more information on the Office of Air's Small Engine Buy-back Program see U.S. EPA (2006c).

22 See Sterner (2003) for a more detailed description of this and other examples of tax-subsidy combinations.

23 For example, Arnold (1995) analyzes the merits of a deposit-refund system in a case study focusing on enhancing used-oil recycling. Sigman (1995) reviews policy options to address lead recycling.

24 See U.S. EPA (2004a), Fisher et al. (1995), and O'Connor (1994).

Because firms have access to better information than the government does, they can measure and report emissions with greater precision and at a potentially lower cost.

Disadvantages of the combined tax-subsidy system may include potentially high implementation and administrative costs, and the political incentive to set the tax too low to induce proper behavior (a danger with any tax). Policy makers may adjust an emissions tax to account for temporal variation in marginal environmental damages, but a tax on output sold in the market cannot be matched temporally or spatially to emissions during production. In addition, to the extent that emissions (e.g., $SO_2$ from power plants) are easily and accurately monitored, other market incentives may be more appropriate. If a production process has many different inputs with different contributions to environmental damages, then it is necessary to tax the inputs at different rates to achieve efficiency. Likewise, if firms are heterogeneous and select a different set of clean inputs or abatement options based on firm-specific cost considerations, then the subsidy should be adjusted for differences in these production functions.[25] A uniform subsidy combined with an output tax may be a good proxy, however, when there is limited heterogeneity across inputs' contribution to emissions and across firms.

Conceptually similar to the tax-subsidy combination is the requirement that firms post performance bonds that are forfeited in the event of damages, or that firms contribute up-front funds to a pool. Such funds may be used to compensate victims in the event that proper environmental management of a site for natural resource extraction does not occur. To the extent that the company demonstrates it has fulfilled certain environmental management or reclamation obligations, the deposited funds are usually refunded. Financial assurance requirements have been used to manage closure and post-closure care for hazardous waste treatment, storage, and disposal facilities. Performance bonds have also been required in extraction industries such as mining, timber, coal, and oil.[26]

## 4.4 Other Market-Oriented or Hybrid Approaches

In addition to the four classic market-based instruments discussed above, several other market-oriented approaches are often discussed in the literature and are increasingly used in practice. Often, these approaches combine aspects of command-and-control and market-based incentive policies. As such, they do not always present the most economically efficient approach. Either the level of abatement or the cost of the policy is likely to be greater than what would be achieved through the use of a pure market-based incentive approach. Nevertheless, such approaches are appealing to policy makers because they often combine the certainty associated with a given emissions standard with the flexibility of allowing firms to pursue the least costly abatement method. This section discusses the following market-oriented approaches:

- Combining standards and pricing approaches;
- Liability rules; and
- Information as regulation.

### 4.4.1 Combining Standards and Pricing Approaches

Pollution standards set specific emissions limits, thereby reducing the probability of excessively high damages to health or the environment. Such standards may impose large costs on polluters. Emissions taxes restrict costs by allowing polluters to pay a tax on the amount they emit rather than undertake excessively expensive abatement. Taxes, however, do not set a limit on emissions, and leave open the possibility that pollution may be excessively high. Some researchers suggest a policy that limits both costs and pollution, referred to as a "safety-valve" approach to regulation, which combines standards with pricing mechanisms.[27] In the case of a standard and tax combination, the same emissions standard is imposed on all

25 The main advantages and disadvantages of deposit-refund systems are discussed in U.S. GAO (1990); Palmer, Sigman, and Walls (1997); and Fullerton and Wolverton (2001, 2005).

26 For more information on the use of financial assurance or performance bonds, see Boyd (2002).

27 See Roberts and Spence (1976) and Spence and Weitzman (1978).

polluters and all polluters are subject to a unit tax for emissions in excess of the standard.

While a standard and pricing approach does not necessarily ensure the maximization of social welfare, it can lead to the most cost-effective method of pollution abatement. This policy combination has other attractive features. First, if the standard is set properly, the desired protection of health and the environment will be assured. This feature of the policy maintains the great advantage of a standards approach: protection against excessively damaging pollution levels. Combining approaches allows for more certainty in the expected environmental and health effects of the policy than would occur with a market-based approach alone. Second, high abatement cost polluters can defray costs by paying the emissions fee instead of cleaning up. This feature preserves the flexibility of emissions taxes: overall abatement costs are lower because polluters with low abatement costs reduce pollution while polluters with high abatement costs pay taxes.

### 4.4.2 Information Disclosure

Requiring disclosure of environmental information has been increasingly used as a method of environmental regulation. Disclosure strategies are most likely to work when there is a link between the polluting firm and affected parties such as consumers and workers.[28] Disclosure requirements attempt to minimize inefficiencies in regulation associated with asymmetric information, such as when a firm has more and better information on what and how much it pollutes than is available to the government or the public. By collecting and making such information publicly available, firms, government agencies, and consumers can become better informed about the environmental and human health consequences of their production and consumption decisions. In some cases, the availability of this information may also encourage more environmentally benign activities and discourage environmentally detrimental ones. For example, warning labels on hazardous substances that describe safe-handling procedures or the risks posed by the product may encourage hazardous substance handlers to take greater precautions, and/or may encourage consumers to switch to less damaging substitutes for some or all uses of the substance. Similarly, a community with information on a nearby firm's pollution activity may exert pressure on the firm to reduce emissions, even if formal regulations or monitoring and enforcement are weak or nonexistent.[29]

Requirements for information disclosure need not be tied explicitly to an emissions standard; however, such requirements are consistent with a standard-based approach because the information provided allows a community to easily understand the level of emissions and the polluters' level of compliance with existing standards or expectations. As is the case with market-based instruments, polluters still have the flexibility to respond to community pressure by reducing emissions in the cheapest way possible.

The use of information disclosure or labeling rules has other advantages. When expensive emissions monitoring is required to collect such information, reporting requirements that switch the burden of proof for monitoring and reporting from the government to the firm might result in lower costs, because firms are often in a better position to monitor their own emissions. If accompanied by spot checks to ensure that monitoring equipment functions properly and that firms report results accurately, information disclosure can be an effective form of regulation. Without the appropriate monitoring, however, information disclosure might not result in an efficient outcome.

While information disclosure has its advantages, it is important to keep three caveats in mind when considering this method for environmental regulation. First, the use of information as regulation is not costless: U.S. firms report spending approximately $346 million per year

28 See OMB (2010b) for guidance issued to regulatory agencies on the use of information disclosure and simplification in the regulatory process.

29 For more information on how information disclosure may help to resolve market failures, see Pargal and Wheeler (1996), Tietenberg (1998), Tietenberg and Wheeler (2001), and Brouhle and Khanna (2007).

to monitor and report releases.[30] Any required investments in pollution control are in addition to this amount. Second, the amount of pressure a community exerts on an emitting plant may be related to socioeconomic status. Poorer, less-educated populations tend to exert far less pressure than communities with richer, well-educated populations.[31] Third, information disclosure may not result in a socially efficient level of pollution when consumers either consider only the effect of emissions on them as individuals, ignoring possible ecological or aggregate societal effects, or when they do not understand how to properly interpret the released information in terms of the health risks associated with exposure to particular pollutants.

EPA-led information disclosure efforts include the Toxics Release Inventory (TRI) and the mandatory reporting of greenhouse gases (GHG). Both the TRI and the GHG reporting rule require firms to provide the government and public with information on pollution at each plant, on an annual basis, if emissions exceed a threshold. There are also consumer-based information programs targeting the risks of particular toxic substances, the level of contamination in drinking water, the dangers of pesticides, and air quality index forecasts for more than 300 cities. There is some evidence in the literature regarding the impact of TRI reporting on firm value: the most polluting firms experience small declines in stock prices on the day TRI emission reports are released to the public. Hamilton (1995) finds a stock price return of -0.03 percent due to TRI report release. Firms that experienced the largest drop in their stock prices also reduced their reported emissions by the greatest quantity in subsequent years.[32]

### 4.4.3 Liability Rules

**Liability rules** are legal tools of environmental policy that can be used by victims (or the government) to force polluters to pay for environmental damages after they occur. These instruments serve two main purposes: (1) to create an economic incentive for firms to incorporate careful environmental management and the potential cost of environmental damages into their decision-making processes; and (2) to compensate victims when careful planning does not occur. These rules are used to guide courts in compensation decisions when the court rules in favor of the victim. Liability rules can serve as an incentive to polluters. To the extent that polluters are aware that they will be held liable before the polluting event occurs, they may minimize or prevent involvement in activities that inflict damages on others. In designing a liability rule it is important to evaluate whether damages depend only on the amount of care taken on the part of the polluter or also on the level of output; and whether damages are only determined by polluter actions or are also dependent on the behavior of victims. For instance, if victims do not demonstrate some standard of care in an attempt to avoid damages, the polluter may not be held liable for the full amount. If damages depend on these other factors in addition to polluter actions, then the liability rule should be designed to provide adequate incentives to address these other factors.

While a liability rule can be constructed to mimic an efficient market solution in certain cases, there are reasons to expect that this efficiency may not be achieved. First, uncertainty exists as to the magnitude of payment. The amount that polluters are required to pay after damages have occurred is dependent on the legal system and may be limited by an inability to prove the full extent of damages or by the ability of the firm to pay. Second, liability rules can generate relatively large costs, both in terms of assessing the environmental damage caused, and the damages paid.[33] Thus, liability rules are most useful in cases where damages requiring compensation are expected to be stochastic (e.g., accidental releases), and where monitoring firm compliance with regulatory procedures is

30 See O'Connor (1996) for information on the costs of monitoring and reporting environmental information. See World Bank (2000) for a discussion of the main advantages and disadvantages of information disclosure as a policy tool.

31 See Hamilton (1993), and Arora and Cason (1999).

32 Hamilton (1995); Konar and Cohen (1997); and Khanna, Quimio, and Bojilova (1998) are empirical studies that have investigated how the TRI has affected firm behavior and stock market valuation.

33 See Segerson (1995), and Alberini and Austin (2001) for discussions of the types of liability rules, the efficiency properties of each type of rule, and an extensive bibliography.

difficult. Depending on the likely effectiveness of liability rules to provide incentives to firms to avoid damages, they can be thought of as either an alternative to or as a complement to other regulatory approaches.

*Strict liability* and *negligence* are two types of liability rules relevant to polluters. Under strict liability, polluters are held responsible for all health and environmental damage caused by their pollution, regardless of actions taken to prevent the damages. Under negligence, polluters are liable only if they do not exhibit "due standard of care." Regulations that impose strict liability on polluters may reduce the transactions costs of legal actions brought by affected parties. This may induce polluters to alter their behavior and expend resources to reduce their probability of being required to reimburse other parties for pollution damages. For example, they may reduce pollution, dispose of waste products more safely, install pollution control devices, reduce output, or invest in added legal counsel.

Liability rules have been used in the remediation of contaminated sites under the Comprehensive Environmental Response, Compensation and Liability Act (CERCLA), also known as *Superfund*, and under the Corrective Action provisions of the Resource Conservation and Recovery Act (RCRA). These rules have also been used in the redevelopment of potentially contaminated industrial sites, known as *brownfields*.

## 4.5 Selecting the Appropriate Market-Based Incentive or Hybrid Approach

Selection of the most appropriate market-based incentive or hybrid regulatory approach depends on a wide variety of factors, including: [34]

- The type of market failure being addressed;
- The specific nature of the environmental problem;
- The type of pollutant information that is available and observable;
- The degree of uncertainty surrounding costs and benefits;
- Concerns regarding market competitiveness;
- Monitoring and enforcement issues;
- Potential for exacerbating economy-wide distortions; and
- The ultimate goals of policy makers.

### 4.5.1 The Type of Market Failure

There are two main types of market failure that are commonly addressed through the use of market-based or hybrid instruments. The first, externality, occurs when firms or consumers fail to integrate into their decision making the impact of their own production or consumption decisions on entities external to themselves. The second type of market failure, asymmetric information, occurs when firms or consumers are unable to make optimal decisions due to lack of information on available abatement technologies, emission levels, or associated risks. Market-based or hybrid instruments that incorporate the marginal external damages of a unit of pollution into a firm or consumer's cost function address the first type of market failure. Information disclosure or labeling are often suggested when the second type of market failures occurs. As discussed in Section 4.4.2, policy makers believe that private- and public-sector decision makers will act to address an environmental problem once information has been disseminated.

### 4.5.2 The Nature of the Environmental Problem

The use of a particular market-oriented approach is often directly associated with the nature of the environmental problem. Do emissions derive from a point source or a nonpoint source? Do emissions stem from a stock or flow pollutant? Are emissions uniformly mixed or do they vary by location? Does pollution originate from stationary or mobile

34 Helpful references that discuss aspects to consider when comparing among different approaches include Hahn and Stavins (1992), OECD (1994a, 1994b), Portney and Stavins (2000), and Sterner (2003).

sources?[35] Point sources, which emit at identifiable and specific locations, are much easier to control than diffuse and often numerous nonpoint sources, and therefore are often responsive to a wide variety of market instruments. Although nonpoint sources are not regulated under EPA, the pollution emitted from a nonpoint source is. Clearly, this makes the monitoring and control of nonpoint source emissions challenging. In instances where both point and nonpoint sources contribute to a pollution problem, a good case can be made for a tax-subsidy combination or a marketable permit system. Under these alternatives, emissions from point sources might be taxed while nonpoint source controls are subsidized.

Flow pollutants tend to dissipate quickly, and it is possible to rely on a wide variety of market and hybrid instruments for emissions control. But stock pollutants persist in the environment and tend to accumulate over time. Controlling stock pollutants may require strict limits to prevent bioaccumulation or detrimental health effects at small doses, making direct regulation a potentially more appealing approach. If these limits are not close to zero, then potentially practical instrument options include a standard-and-pricing approach or a marketable permit approach that defines particular trading ratios to ensure that emission standards are not violated at any given source are. These same instruments are appealing when pollutants are not uniformly mixed across space. In the case of non-uniformly mixed emissions, it is important to account for differences in baseline pollution levels, and differences in emissions across more and less polluted areas.

Stationary sources of pollution are easier to identify and control through a variety of market instruments than are mobile sources. Highly mobile sources are usually numerous, each emitting a small amount of pollution. Emissions therefore vary by location and damages can vary by time of day or season. For example, health impacts associated with vehicle traffic are primarily a problem at rush hour when roads are congested and cars spend time idling or in stop-and-go traffic. Differential pricing of resources used by these mobile sources (such as higher tolls on roads or greater subsidies to public transportation during rush hour) is a potentially useful tool.

35 For a detailed discussion of how the nature of the environmental problem affects instrument choice, see Kahn (1998), Goulder et al. (1999), Parry and Williams (1999), Harris (2002), Tietenberg (2002), and Sterner (2003).

### 4.5.3 The Type of Pollutant Information that is Available and Observable

The selection of market-oriented approach may depend on the available data. Is the level of pollutant actually observable or measurable? Or will the level need to be imputed based on inputs and technology used? Are the sources heterogeneous? Does the pollutant vary across time and space? Are information technologies available to the analyst to improve data collection? When the pollutant concentration can be directly and easily measured then it is possible to directly regulate the level of the pollutant. But if monitoring costs are high, it may be easier to target a particular input or require a specific technology known to reduce pollutants by a certain amount. The pollutant levels can be imputed based on regulation placed on the input or the technology used.

The link between pollution and heterogeneous sources is often difficult and costly to determine, and costs may increase if the pollutant levels vary over time. Uniform policies are often used for the sake of simplicity. However, information technologies such as continuous emissions monitoring equipment (CEMs) or geographical information systems (GIS) can be used to link sources to pollutant levels. In these cases, policies that make use of this new information may be used and often can reduce costs. As technology improves or more data become available, analysts should consider reassessing the regulation design.[36]

### 4.5.4 Uncertainty in Abatement Costs or Damages

The choice between price-based instruments (e.g., taxes or charges) and quantity-based

36 For more information see Xabadia, Goetz, and Zilberman (2008).

instruments (e.g., marketable permits) has been shown theoretically to rest on the uncertainty surrounding estimated benefits and costs of pollution control, as well as on how marginal benefits and costs change with the stringency of the pollution control target. If uncertainty associated with the cost of abatement exists but damages do not change much with additional pollution, then policy makers can effectively limit costs by using a price instrument without having much impact on the benefits of the policy. If, on the other hand, there is more uncertainty associated with the benefits of controlling pollution and policy makers wish to guard against high environmental damages, a quantity instrument is likely preferable.[37] In this way, the policy maker can avoid potentially costly or damaging mistakes. The policy maker should also be aware of any discontinuities or threshold values above which sudden large changes in damages or costs could occur in response to a small increase in the required abatement level.

### 4.5.5 Market Competitiveness

Market power is a type of market failure in and of itself, as it may result in output that is too low and prices that are too high compared to what would occur in a competitive market. Instruments that cause firms to further restrict output may create additional inefficiencies in sectors where firms have some degree of market power. A combination of market-based instruments may work more effectively than a single instrument in this instance. To the extent that cost burdens are differentiated, the use of certain market-based instruments may cause a change in market structure that favors existing firms by creating barriers of entry and allowing existing firms a certain amount of control over price. Permit systems that set aside a certain number of permits for new firms, for instance, may guard against such barriers.

### 4.5.6 Monitoring and Enforcement Issues

Market-oriented instruments differ in the degree of effort required to monitor and enforce the desired emissions level. For example, subsidies, deposit-refund systems, and information disclosure shift the burden of proof to demonstrate compliance from government to the regulated entities. Because firms are generally in a better position than government to monitor and report their own emissions, they likely can do so at a potentially lower cost. This feature makes such approaches attractive when monitoring is difficult or emissions must be estimated (e.g., when there are nonpoint sources or large numbers of small polluters). In these cases, attempts to prohibit or tax the actions of polluters are likely to fail due to the risk of widespread noncompliance (e.g., illegal dumping to avoid the tax) and costly enforcement.

### 4.5.7 Potential for Economy-Wide Distortions

Analysts should consider the potential distortionary effects of any policy option considered. Even if a policy is deemed relatively efficient on its own, it may interact with pre-existing environmental, economic, or agricultural policies (e.g., product standards, non-environmental subsidies, trade barriers) in non-intuitive ways that can exacerbate distortions in the economy and result in unintended environmental consequences. Instruments that include a revenue-raising component, such as auctioned permits or taxes, may allow for opportunities to direct collected resources to reduce other taxes and fees and the associated inefficiencies.[38] See Chapter 8 and Appendix A for a more detailed discussion of economy-wide distortions.

37 See Weitzman (1974) for the classic paper on the ways in which uncertainty (also referred to as lack of information) affects instrument choice. See Chapter 10 of these *Guidelines* for more information on the treatment of uncertainty in analyses.

38 For useful references on the issues concerning the uses of revenues from pollution charges (e.g., applying environmental tax revenues so as to reduce other taxes and fees in the economy) and ways to analyze these policies, see Bovenberg and de Moojii (1994), Goulder (1995), Bovenberg and Goulder (1996), Goulder et al. (1997), and Jorgenson (1998a, 1998b).

### 4.5.8 The Goals of the Policy Maker

Finally, the goals of policy makers may influence the instrument selected to regulate pollution. Each considered instrument may have different distributional and equity implications for both costs and benefits; these implications should be accounted for when deciding among instruments. For example, policy makers may wish to ensure clean-up of future pollution by firms. Policy makers may consider using insurance and financial assurance mechanisms to supplement existing standards and rules when there is a significant risk that sources of future pollution might be incapable of financing the required pollution control or damage mitigation method. In addition, the degree to which policy makers want to allow the market to determine exact outcomes may influence the choice of instrument. The quantity of marketable permits issued, for example, sets the total level of pollution control, but the market determines which polluters reduce emissions. On the other hand, taxes let the market determine both the extent of control by individual polluters and the total level of control.

## 4.6 Non-Regulatory Approaches

EPA has pursued a number of non-regulatory approaches that rely on **voluntary initiatives** to achieve emissions reductions and improve management of environmental hazards. These programs are usually not intended as substitutes for formal regulation, but instead act as important complements to existing regulation. Many of EPA's voluntary programs encourage polluting entities to go beyond what is mandated by existing regulation. Other voluntary programs have been developed to improve environmental quality in areas that policy makers expect may be regulated in the future but are currently not regulated, such as GHG emissions and nonpoint source water pollution.[39]

Much of the technical foundation for these voluntary initiatives rests on the concepts underlying a "pollution prevention" approach to environmental management choices. In the Pollution Prevention Act of 1990, Congress established a national policy that:

- Pollution should be prevented or reduced at the source whenever feasible;
- Pollution that cannot be prevented should be recycled in an environmentally safe manner whenever feasible;
- Pollution that cannot be prevented or recycled should be treated in an environmentally safe manner whenever feasible; and
- Disposal or other release into the environmental should be employed as a last resort and should be conducted in an environmentally safe manner.

EPA typically designs its voluntary programs through regular consultation (but little direct negotiation) with affected industries or consumers.[40] In many cases, voluntary programs facilitate problem solving between EPA and industry because information on procedures or practices that reduce or eliminate the generation of pollutants and waste at the source are shared through the consultative process.

In slightly more than a decade, voluntary programs at EPA have increased from two programs to approximately 40 programs involving more than 13,000 organizations. Partner organizations include small and large businesses, citizen groups, state and local governments, universities, and trade associations.[41] Voluntary programs in which these groups participate tend to have either broad environmental objectives targeting a variety of firms from different industries, or focus on more specific environmental problems relevant to a single industrial sector. In the United States, nearly

39 While this chapter only discusses government-led voluntary initiatives at the federal level at EPA, other government agencies, industry, non-profits, and international organizations have also initiated and organized voluntary initiatives designed to address particular environmental issues. These initiatives are beyond the scope of this chapter, which limits itself to a brief description of policy options available to EPA.

40 Because these programs are voluntary there is no need for formal public comment. However, industry often is consulted during the design phase.

41 For information on EPA's voluntary programs, see the Partners for the Environment List of Programs at http://www.epa.gov/partners/programs/index.htm (accessed November 03, 2010) (U.S. EPA 2008e).

one third of all multi-sector federal voluntary programs focus on energy efficiency and climate change issues. General pollution prevention efforts represent the next most popular type of voluntary program. Single-sector federal voluntary programs tend to target environmental problems associated with transportation-related issues and energy producing sectors such as coal mining and power generation. These programs strive to provide participating firms with targeted and effective technological expertise and assistance.[42]

## 4.6.1 How Voluntary Approaches Work

Voluntary programs can use the following four general methods to achieve environmental improvements: (1) require firms or facilities to set specific environmental goals; (2) promote firm environmental awareness and encourage process change; (3) publicly recognize firm participation; and (4) use labeling to identify environmentally responsible products. These methods are not mutually exclusive, and most U.S. voluntary programs use a combination of methods.

Goal setting is a very common method used in the design of voluntary programs. Implementation-based goals are typically EPA-specified, program-wide targets designed to provide a consistent objective across firms. Target-based goals are usually qualitative and process-oriented so that firms may individually set a unique target. EPA's WasteWise and Climate Challenge programs are examples of programs with target-based goals. EPA's 33/50 program, which set a goal of a 33 percent reduction of toxic emissions by firms in the chemical industry by 1992, and a 50 percent reduction by 1995 (relative to a 1988 baseline), is an example of a voluntary program with an implementation-based goal.

Programs designed to promote environmental awareness and to encourage process change within firms often involve implementing a system to evaluate firms' ongoing operations and to provide information on newly available technologies. Examples of this type of approach include the SmartWay program, which encourages firms to adopt energy efficient changes that also yield fuel savings for freight trucking companies, and the Green Suppliers Network program, which provides partner firms with technical reviews and suggestions on how to eliminate waste from production processes.

Voluntary programs that publicly recognize firm participation are designed to provide green consumers and investors with new information that may alter their consumption and investment patterns in favor of cleaner firms. Firms may also use their environmental achievements to differentiate their products from competitors' products.[43] These information and firm differentiation effects are the intent of the Green Power Partnership and the WasteWise program.

Finally, product labeling can be applied to either intermediate inputs in a production process or to a final good. Labels on intermediate goods encourage firms to purchase environmentally responsible inputs. Labels on final goods allow consumers to identify goods produced using a relatively clean production process. For example, products deemed by EPA to be energy efficient may be eligible for the Energy Star or Design for the Environment labels.

## 4.6.2 Economic Evaluation of Voluntary Approaches

A formal economic analysis is not required for the selection and implementation of a non-regulatory or voluntary approach to pollution reduction.

Several factors contribute to the difficulty of evaluating voluntary approaches. Many programs target general environmental objectives and thus

42 See Khanna (2001); OECD (1999, 2003); U.S. EPA (2002a); and Brouhle, Griffiths, and Wolverton (2005) for discussions of how voluntary programs work and how they are used in U.S. environmental policy making.

43 See Arora and Cason (1995); Arora and Gangopadhyay (1995); Konar and Cohen (1997, 2001); Videras and Alberini (2000); Brouhle, Griffiths, and Wolverton (2005); and Morgenstern and Pizer (2007) for more information on the main arguments for why firms participate in voluntary programs.

### Text Box 4.3 - Water Quality Trading of Nonpoint Sources

In 2003, EPA issued a "Water Quality Trading Policy" (U.S. EPA 2003d) that encourages states and tribes to develop and implement voluntary water-quality trading to control nutrients and sediments in areas where it is possible to achieve these reductions at lower costs. Under the Clean Water Act, EPA is required to establish Total Maximum Daily Loadings (TMDL) of pollutants for impaired water bodies. The TMDL does not establish an aggregate cap on discharges to the watershed, but it does provide a method for allocating pollutant discharges among point and nonpoint sources. Point sources are regulated by EPA and, as such, are required to hold National Pollutant Discharge Elimination System (NPDES) permits that limit discharges. However, many water bodies are still threatened by pollution from unregulated, nonpoint sources. Nutrients and sediment from urban and agricultural runoff have led to water quality problems that limit recreational uses of rivers, lakes, and streams; that create hypoxia in the Gulf of Mexico; and that decrease fish populations in the Chesapeake Bay. The impetus for allowing effluent trading between point and nonpoint sources is to lower nutrient and sediment loadings and to improve or preserve water quality.

To ensure that the reduction resulting from the trade has the same effect on the water quality as the reduction that would be required without the trade, trading ratios are often applied. These ratios attempt to control for the differential effects resulting from a variety of factors, which may include:

- location of the sources in the watershed relative to the downstream area of concern;
- distance between the permit buyer and seller;
- uncertainty about nonpoint source reductions;
- equivalency of different forms of the same pollutant discharged by the trading partners; and
- additional water quality improvements above and beyond those required by regulation.

The idea behind trading is to allow point sources to meet the discharge limit at a lower cost. This allows continued growth and expansion of production, while giving nonpoint sources an incentive to reduce pollution through participation in the market. To the extent that it is cheaper for a nonpoint source to reduce pollution than to forgo revenues earned from the sale of any unused credits to point sources, the nonpoint source is predicted to choose to emit less pollution.

As of March 2007, 98 NPDES permits, covering 363 dischargers, included provisions for trading. However, only about a third of the dischargers had carried out one or more trades under these permits (U.S. EPA 2007f). Trading has been limited for several reasons. First, there is no aggregate "cap" on discharges that applies to both point and nonpoint sources within a watershed. Reductions by nonpoint sources are essentially voluntary. Point-source dischargers often explore trading as a way to expand production while meeting the requirements of their individual permits, but there is no general signal in the market to do so. Second, these are often thin markets. The way in which the market is designed or trading ratios are established can make it difficult or expensive for an entity to identify and complete a trade. Third, while Best Management Practices (BMPs) are typically used to define a pollution reduction credit from a nonpoint source, uncertain or changing climatic conditions, river flow, and stream conditions make it difficult to measure the effect of a BMP on water quality. Such uncertainty also makes measuring and enforcing a pollution reduction from a nonpoint source difficult. Fourth, encouraging nonpoint source involvement in trading, given the agriculture industry's distrust of regulators, is challenging. Finally, it is difficult to define appropriate trading ratios between point and nonpoint sources.

lack a measurable environmental outcome. Even if a measurable output exists, there may be a lack of data on a firm's or industry's environmental outputs. In order to perform an evaluation, a reasonable baseline from which to make a comparison must be established. This requires an extensive analysis comparing the actions of participants to non-participants in the program;

such data is likely difficult and costly to obtain.[44] Any economic evaluation of voluntary programs should net out pollution abatement activities that would have occurred even if the voluntary program were not in place. Some of these evaluation obstacles can be overcome if voluntary approaches use more defined and detailed goal setting and require more complete data collection and reporting from the outset.[45]

The economic literature evaluating the efficacy of voluntary programs is decidedly mixed. The vast majority of existing empirical studies focus on a few large, multi-sector voluntary programs such as 33/50, Green Lights, and Energy Star. For these programs, there is some evidence of success in reducing participant emissions. However, studies generally fail to account for non-program factors such as the ability to count reductions that occurred prior to the start of the program; to compare reductions relative to a baseline counterfactual may overstate these reductions. Researchers have been less successful in demonstrating that voluntary programs have led to greater emission reductions than would have occurred without the program in place. One thread of literature points to the positive impact of a regulatory threat on voluntary program effectiveness. When the threat of regulation is weak, abatement levels are likely to be lower. However, when the threat of regulation is strong, levels achieved are closer to those under optimal regulatory action.

## 4.7 Measuring the Effectiveness of Regulatory or Non-Regulatory Approaches

Several policy criteria should be considered when evaluating the success of regulatory or non-regulatory approaches. These include environmental effectiveness; economic efficiency; savings in administrative, monitoring and enforcement costs; inducement of innovation; and increased environmental awareness. In many cases, analysis of these factors will make evident the particular advantages of one or more market-based incentive approaches over command-and-control regulation. While a formal analysis may not be required when considering the implementation of a non-regulatory approach, these factors are still important to consider. According to recent reviews (Harrington et al. 2004, and Goulder and Parry 2008) it is unlikely that any one policy will dominate on all of these factors. However, in many areas an incentive policy, if available, can be more cost-effective than a competing command-and-control policy.

In determining the effectiveness of a policy approach, policy makers should consider the following factors and questions:

- **Environmental Effectiveness:** Does the policy instrument accomplish a measurable environmental goal? Does the policy instrument result in general environmental improvements or emission reductions? Does the approach induce firms to reduce emissions by greater amounts than they would have in the absence of the policy?
- **Economic Efficiency:** How closely does the approach approximate the most efficient outcome? Does the policy instrument reach the environmental goal at the lowest possible cost to firms and consumers?
- **Reductions in Administrative, Monitoring, and Enforcement Costs:** Does the government benefit from reductions in costs? How large are these cost savings compared to those afforded by other forms of regulation?
- **Environmental Awareness and Attitudinal Changes:** In the course of meeting particular goals, are firms educating themselves on the nature of the environmental problem and ways in which it can be mitigated? Does the promotion of firm participation or compliance affect consumers' environmental awareness or priorities and result in a demand for greater emissions reductions?
- **Inducement of Innovation:** Does the policy instrument lead to innovation in

44 See Chapter 5 for a discussion of baselines and specifically Section 5.7 for a discussion of behavioral responses.

45 See Segerson and Miceli (1998); Khanna and Damon (1999); National Research Council (2002); Segerson and Wu (2006); Morgenstern and Pizer (2007); and Brouhle, Griffiths, and Wolverton (2009).

abatement techniques that decrease the cost of compliance with environmental regulations over time?

To address a number of these key evaluation criteria, *Guidelines* Chapters 8 and 9 offer instruction on how to measure social costs and how to address equity issues, respectively.