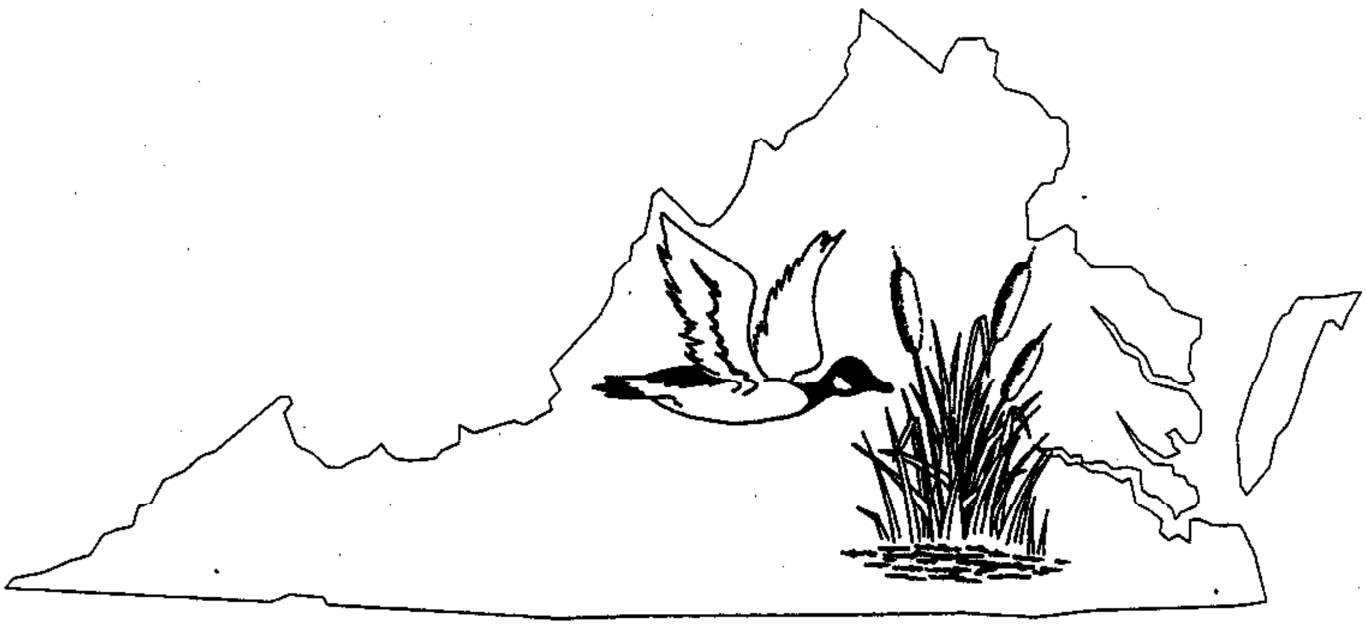
AN INTRODUCTION TO MARKET-BASED STRATEGIES FOR CHESAPEAKE BAY POLICY AND MANAGEMENT



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THE PROBLEM

One of the most persistent and extensive threats to water quality in the Chesapeake Bay and its tributaries is excessive nutrient loadings. Nitrogen and phosphorous are the primary cause of excessive algae growth in Bay waters. By blocking sunlight penetration through the water, algae blooms adversely effect the growth of submerged aquatic vegetation (SAV) and thus degrade important fishery habitat. When the algae dies, decay and decomposition deplete the oxygen content of the water. Consequently, some areas of the Bay are completely void of dissolved oxygen (Virginia Department of Environmental Quality 1993).

Nitrogen and phosphorous enter Bay waters from a variety of sources. The majority of the nutrients entering the Bay - three-quarters of all nitrogen and a two-thirds of the phosphorous - originate from nonpoint sources (Chesapeake Bay Program 1993). Nonpoint nutrient discharges arise from many different sources - forestry and agricultural operations (crop and livestock operations) and urban/suburban area runoff. The largest contributor of nonpoint nutrient pollution is agriculture. Point sources account for the remaining nitrogen and phosphorous entering the Bay. Municipal waste water treatment plants contribute the vast majority of the point source nutrients (Chesapeake Bay Program 1993).

In response to these problems Maryland, Pennsylvania, Virginia, the District of Columbia, the Chesapeake Bay Commission, and the Environmental Protection Agency established a 40 percent reduction goal for controllable nitrogen and phosphorous loads at the main stem of the Bay by the year 2000. Under current regulations, municipal and industrial point sources face technology based, enforceable discharge limits. Discharge limitations are imposed through state and federal permitting processes. Some large confined livestock operations are also covered by the federal permitting program. The control of nonpoint sources of nutrient pollution has taken a different course. Nonpoint pollution control has relied primarily on a combination of educational efforts and government subsidies to induce reductions in nutrient pollution.

While there have been significant efforts devoted to reducing the level of nutrients entering the Chesapeake Bay since the mid-1980s, significant reductions still need to be made in order to achieve the 40 percent reduction goal. Between 1984 and 1992, total phosphorous concentrations have dropped by 16 percent and nitrogen levels have remained virtually unchanged (Chesapeake Bay Program 1993). If the nutrient reduction goal in the Chesapeake Bay is to be achieved by the target date of 2000, new and innovative approaches to manage the nutrient pollution problem will need to be employed.

Toxic contaminants also pose a potential threat to achieving water quality goals in the Bay. In 1994 the Chesapeake Executive Council (1994, 3) approved an environmental goal related to toxics in the Chesapeake Bay:

> Our goal is a Chesapeake Bay *free from toxic impacts*. We will work towards this goal by reducing or eliminating the input of chemical contaminants from all controllable sources to levels that result in no toxic or bioaccumulative impact on the living resources that

inhabit the Bay or on human health (italics added).

A toxics reevaluation in 1992 found no evidence of severe, systemwide responses to chemical contaminants similar in magnitude to the observed effects throughout the Bay due to excessive nutrient levels. In most areas, chemical contaminants are below thresholds associated with adverse impacts on the Bay's living resources, but elevated above natural background levels. The long-term effects from these low levels remain unclear.

While overall levels of toxics in the Bay do not appear to pose an immediate threat to living resources, there are several areas in the Bay that are adversely impacted by toxic contaminants. Chemical contaminants in sediment deposits in the Baltimore Harbor, Back River, Anacostia River and Elizabeth River are all high enough to adversely impact aquatic life (Chesapeake Bay Program 1994). Furthermore, given the prevalent use of toxic substances in a vast array of industrial, agricultural, and household uses, there are always environmental risks associated with accidental spills or illegal dumpings.

While progress has been made in the environmental restoration of the Chesapeake Bay, rapid development pressure and population growth throughout the Bay region will place new demands and stress on Bay resources. While there is a documented need to make further progress in improving the environmental health of the Bay, it is also being increasingly recognized that further efforts to improve environmental quality will become progressively more costly to the public and private sectors. This growing concern with the costs of environmental protection has coincided with increasingly common complaints that many existing environmental regulations are too inflexible and insensitive to individual circumstances and choices. One of the great challenges we face is to devise strategies and mechanisms to maintain a balance between the inevitable growth and development in the Bay watershed and the ecological health of the Bay.

Market-based environmental policies

The two goals of improved environmental quality and more flexible, cost-effective environmental policies are not always in direct opposition to each other. In short, environmental policy can be accomplished in a more cost-effective fashion, allowing for more individual discretion in making choices related to the environment, while at the same time *increasing and improving* environmental protection. A set of policy tools that can be used to better achieve these dual objectives are *market-based* or *incentive-based* environmental policies. The development and use of these policies can help build common policy ground between the proponents of environmental protection, economic growth, and individual choice (Hahn and Stavins 1991; Hahn 1989).

Market-based environmental policies achieve socially desirable environmental goals and objectives by taking advantage of two characteristics of markets: price signals and individual choice. The basic idea behind all market-based tools is to create a financial incentive for individuals to change behavior by placing a cost or price on that behavior. By rewarding favorable behavior or by placing costs on undesirable activities, market-based tools redirect the incentives of individuals toward making choices consistent with environmental objectives. One proponent writes these "policies force consumers and producers to experience the full costs and consequences of their decisions; not afterwards when its too late to affect their decisions, but beforehand, when it still makes a difference" (Stavins 1991, 2). This factor is consistent with desires to emphasize pollution prevention over treatment. Because market-based policies rely on the same financial incentives found in a the marketplace, these policies can also be described as incentive-based approaches.

If imposing a price (cost) on environmental undesirable activity *provides the incentive* to change behavior, then greater individual discretion in making choices *provides the mechanism* to respond to these price signals. In markets for everyday goods like clothes and food, markets solve the tasks of allocating goods and services between millions of businesses and people by providing a framework capable of utilizing the information revealed in countless individual decisions. Markets work precisely because they allow people to respond in their own best personal interest given their unique economic circumstances. Market-based environmental policies function in an analogous fashion by providing a framework for allowing people the flexibility to determine the best way to comply with socially determined environmental goals. In this way individual decisions also provide the greatest benefit to society.

One of the most important reasons to consider market-based policies is to facilitate the development of innovative pollution control strategies. Conventional air and water pollution control policies, for instance, require specific technological controls or uniform performance standards be applied to large, identifiable pollution sources. The development of more innovative ways to control pollution, however, is muted because the financial incentive to reduce pollution ends once the regulatory requirement has been met (Kneese and Shultze 1978). By contrast, a market-based approach creates a constant incentive to search for ways to reduce pollution. In order to increase their profit position, firms continuously engage in a search to reduce the costs of production. A market-based pollution control strategy seeks to harness these behavioral tendencies by placing a cost on every unit of pollution generated. If pollution control costs can be effectively incorporated into the calculation of production costs, the firm will seek to reduce costs through the continuous search and development of innovative pollution reduction strategies.

There are many different ways in which the price or cost of environmental activities can be established. In general, however, market-based policies can be grouped in three broad categories: 1) fee (charge or subsidy) systems, 2) market development, and 3) liability systems. First, the price of a waste-generating activity or natural resource use can be changed directly through the use of charges or subsidies. In the case of pollution, a charge or tax on each unit of emission or effluent discharged (or a related activity) works to reduce pollution by increasing the price (cost) of polluting behavior. If a person or firm has to pay whenever pollutants are discharged into the environment, a constant incentive is created for the polluter to reduce the amount of effluent released. There are many different examples of these types of policies including emission taxes, front-end taxes, product charges, and deposit-refund systems (Stavins et al. 1988, 1991). Equivalently, subsidies provide financial rewards for undertaking environmentally beneficial activities.

The second general type of market-based environmental policy is the creation or development of a market. Market development involves establishing a structure that allows willing buyers and sellers to come together in voluntary exchange to achieve environmental goals. From the development of such markets, a market price emerges. This price represents a cost to those involved in environmentally harmful activities and a reward for those engaged in producing environmentally friendly outcomes. There are many different ways to develop environmental markets including creation of a tradable permit or quota system and facilitation of environmental markets.

The *third* type of market-based environmental policy is the establishment of liability systems. A system of liability rules assigns financial responsibility to parties whose actions could impose harm or damage on third parties. Although compensatory damages are not paid until harm has occurred, the potential for environmental harms creates incentives for those held potentially liable to undertake preventative measures rather than paying for the damages they cause. Thus, liability places a cost on those engaging in environmentally risky activities by forcing these parties to account for the risk imposed on others.

While these are some of the most important reasons to consider incentive-based policies, we do not argue they are a panacea for every type of environmental and natural resource problem. Some types of environmental problems such as management of highly toxic materials or nuclear wastes may not be well-suited for a market-based approach and a more conventional regulatory strategy may be the more appropriate approach. In the following discussion, efforts will be made to point out the limitations as well as advantages of incentive-based policies. Yet, the primary thrust is that market-based policies are [redacted] considered a set of policy tools that can make an important contribution to improving the environmental quality of the Chesapeake Bay.

MARKET-BASED POLICIES FOR NUTRIENTS AND TOXICS

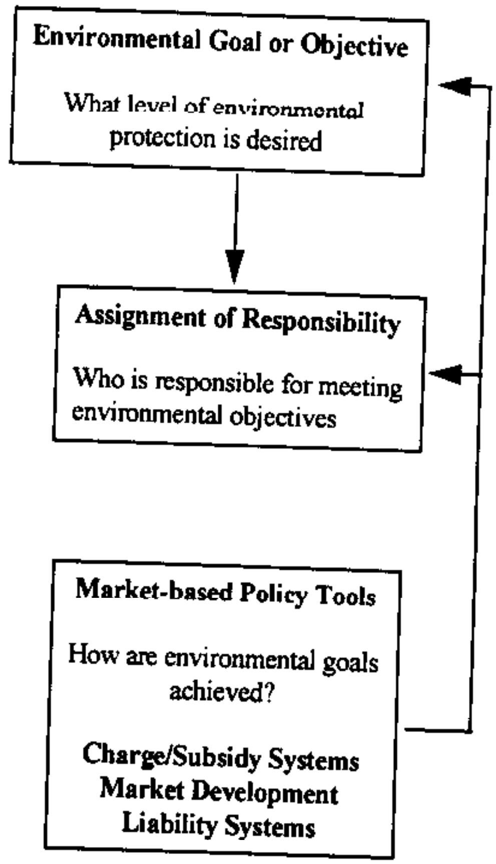
To many people, the idea of market-based environmental policies and environmental protection at first may seem contradictory. Skeptics may ask "Wasn't it a 'free' market that caused many environmental problems to begin with?" Others may object to these policies on moral grounds as "putting a dollar value on the environment." Yet, these are both erroneous objections. The use of market-based environmental policies does not mean turning over the environment to a free (unregulated) market nor does it mean allocating resources according to dollar estimates of environmental amenities.

The role of market-based approaches in the environmental policy process is shown in Figure 1. Economists recognize the vital services that a healthy environment provides. Besides providing fundamental life-sustaining services, a healthy environment also provides people with aesthetic and recreational services. While almost everyone acknowledges the importance of environmental quality, maintaining or improving environmental quality is not a costless undertaking. For instance, achieving a near pollution free environment would be exceedingly expensive, requiring the literal shutdown of entire sectors of the economy. Given that environmental quality is expensive, one arrives at the first difficult set of questions to be addressed in environmental policy: "How clean is clean?" and "How much clean is enough?" (see Figure 1). The answer to this set of questions establishes our environmental goals. It should be stressed that market-based policies do not purport to fully provide a solution to this fundamental social choice.[1] Since the answer to this question involves fundamental social values and choices between competing social objectives, the setting of environmental goals is considered here a question best addressed by political processes (Stewart 1988; Ackerman and Stewart 1985; 1988). The 40 percent reduction in nutrients entering the Chesapeake Bay and the federal no-net-loss of remaining wetlands are both examples of outcomes of environmental goal setting in the political process.

Once the environmental goals have been established, responsibility for achieving these goals will also need to be established (see Figure 1). Assigning responsibility for reaching an environmental goal also means assigning financial responsibility. Market-based policies do not assign this responsibility, but the choice of what type of market-based policies to implement is dependent on this decision. For instance, if it is decided the polluter should bear the responsibility of reducing effluent discharge, then a pollution tax and not a subsidy would be the relevant policy. Thus, an effluent charge is based on the "polluter pays" principle. Obviously, assigning responsibility for achieving environmental goals is ultimately based on the beliefs of members of society of what is considered fair and equitable.

[1] Some economists, however, do argue that selecting the level of environmental quality should be determined according to benefit-cost criteria. As Figure 1 shows, whether environmental goals are established through political process or by economic analysis is a separate issue from the decision to implement market-based environmental policies.

Figure 1: The Role of Market-based Environmental Policy



The remaining question is "How are we to best achieve our environmental goals?" Market-based policies listed in Figure 1 are a group of policy tools -- fee (charge/subsidy) systems, market development, and liability systems -- that provide one set of answers to this question. These policy tools are alternative *means* for achieving environmental goals (Ackerman and Stewart 1988). Thus, market-based tools are not mechanisms to circumvent environmental goals, but rather are tools to help achieve environmental goals consistent with the public's environmental values. In the absence of meaningful environmental goals or assignment of responsibility, market-based mechanisms would be less effective. Economists argue, however, market-based policies offer many improvements over conventional "command and control" regulations as a way of reaching the same environmental goals.

Far from being "free" the implementation of market-based environmental policies requires a government commitment of resources and effort. As the discussion below will make clear, market-based policies operate effectively when government provides a legal and regulatory structure that channels financial incentives or disincentives into environmentally friendly avenues. Obviously, there are implementation and compliance costs to both the private and public sectors associated with these types of policies. As the cost effectiveness of alternative environmental policies becomes clear, environmental goals will be reconsidered or perhaps those responsible for achieving these objectives will shift as perceptions of fairness change. The effectiveness of market-based policies are represented in Figure 1 as arrows "feeding back" to the choices of environmental goals and assignment of responsibility. The most important advantage of market-based environmental policies may be related to this "feedback" between means and ends: the implementation of flexible, cost-effective policies makes higher levels of environmental protection more appealing affordable.

Changing Price Signals: Fees (Charges and Subsidies)

A market-based system achieves an environmental goal by altering the private cost or benefit of the targeted activity. In the following section a variety of fee-based mechanisms are discussed and these can be grouped into two types - direct and indirect applications (Shabman and Phillips 1991). A direct fee system attempts to *directly* alter the prices and costs of the specific behavior causing the environmental problem. An indirect use of a fee would apply the similar charge or subsidy not directly to the environmental problem, but to activities *contributing* or *related* to the problem. As will be shown below, this distinction between direct and indirect application is important since environmental policies are more environmentally and economically effective the closer the policy targets the specific behavioral pattern causing the problem.

Whether direct or indirect, market-based fee systems all share on thing in common -- they are applied on a *per unit* basis. Only when applied on a per unit basis is an effective, market-type incentive created. Consider a municipality that wishes to promote water conservation. Requiring all households to pay a flat monthly fee for water service is not considered here a market-based fee system. Since households pay the same rate regardless of whether 10 gallons or 10,000 gallons of water are used, a flat fee does not create incentives to conserve water. Instituting a charge on every gallon of water used, however, is considered a market-based charge system since the cost of using water is directly linked to the volume of water used. A per unit charge creates a continuous incentive for households to conserve water.

Direct Charges and Subsidies

A direct charge system imposes a cost directly on the environmentally undesirable activity. Examples of direct charge systems include an emission or effluent tax. An emission or effluent tax (a waste-end tax) is imposed on every unit of pollution released into the environment. Pollution dischargers with the ability to reduce effluent at a cost less than the charge will choose to undertake pollution reduction measures to avoid the additional charges. Conversely, dischargers with higher abatement costs may choose instead to increase total waste emissions and pay the additional tax costs. Since the polluter is allowed to decide how to respond, the polluter will implement cost-effective pollution abatement strategies first. The per unit charge on every unit of pollution establishes for all discharges a constant incentive to search for these least-cost measures. Obviously, the magnitude of the per unit charge determines the total reduction in emissions - the higher the charge the more financially advantageous it is for dischargers to implement pollution abatement measures.

The tax not only provides a financial incentive to reduce pollution, it also generates government revenues from socially undesirable activities. Indeed, from an economic incentives perspective, it is argued that taxing environmentally damaging activities is a better way to raise revenues than taxing productive activities like work, savings, and investment (Repetto, Dower, and Gramlich 1993).

Despite the many advantages, a direct fee system is not a practical option for some types of environmental problems. Applying a charge or subsidy is dependent on being able to quantify and measure the problem of interest on a per unit basis. In many instances this is either technologically or administratively impractical (Barthold 1994; Vickrey 1992). Even if emissions can be easily measured, monitoring and implementation of such a system is not a costless undertaking. As the per unit charge increases, so will monitoring costs since the incentive to avoid the charge increases along with the charge rate. Without adequate monitoring and enforcement, a direct charge may inadvertently increase the incidence of illegal discharges and dumping. Furthermore, direct charge systems are usually met with staunch resistance from those who would bear the tax burden.

In principle, a direct subsidy (tax credits or cost share) could also be used to achieve environmental goals. Analogous to a direct charge, a direct subsidy would pay a discharger for each unit of pollution reduced from some baseline level of emissions. Unlike a direct charge system that is based on the polluter pays principle, a subsidy system assumes the public is responsible for financing improved environmental quality. Instead of generating public revenues, a direct subsidy system would require additional public outlays. Preferential land use assessments act in the same way to provide a subsidy.

Indirect Charges and Subsidies

An input charge (or front-end tax) is a charge (tax) added to an input to the production process that is contributing to the environmental problem instead of directly on the quantity of emissions. Under appropriate conditions, the increase in the cost of using the input causing the environmental harm would encourage the substitution of less harmful inputs. A carbon tax is one example of an input tax that could be used to manage CO_2 emissions. A carbon tax places a per unit charge on the amount of carbon contained in fuel used rather than the actual amount of carbon discharged into the environment.

As an alternative to reducing CO_2 via a carbon tax, an additional charge could be placed on each unit of electricity consumed. A charge placed on the output of a production process can be labeled a product charge. In this case, CO_2 emissions would be reduced since less energy would be consumed - and thus generated - at the higher prices.

A variant of the product charge is a deposit-refund system. A deposit refund system begins with placing a charge on purchase of products that have the potential of creating environmental harm through improper disposal of either residues or the container. The charge is partly or entirely refunded when the purchaser of the product returns the container, residue, or unused product to an approved collection center. A deposit refund provides incentives to properly dispose of waste by imposing a cost in the form of a forfeited deposit. The difference between deposit and the refund should equal to the costs of proper disposal (Stedge and Shabman 1995).

As an alternative to the above charge-based schemes, indirect subsidies could be granted for the use or purchase of a less environmentally damaging input or output. An example of an input subsidy is to pay farmers to switch to less toxic or less leachable pesticides. On the other hand, subsidizing the price of a substitute product would provide a positive financial incentive for people to reduce the purchase of environmentally harmful ones. A subsidy to households who purchase more energy-efficient appliances is one example of an output subsidy.

In general indirect fee-based systems are easier to implement and monitor than direct systems. In many cases, these taxes are easily incorporated within the existing tax structure, minimizing implementation and administration costs. However, indirect fee systems are generally not considered as *environmentally* or *economically* effective as direct charge systems. Indirect charges (or subsidies) target only a part of the production and consumption process related to the environmental problem, while other pollution abatement strategies that may be less costly to implement are not incorporated into pollution abatement decisions. Thus, the cost-effectiveness of indirect charge systems is reduced in relation to the elimination of environmentally compatible strategies (choices) available to meet environmental objectives.

Possible Applications of Fee Systems

Fertilizer Tax: For a number of reasons nonpoint source pollution is a difficult problem to address from a regulatory standpoint. Nonpoint pollution sources are often small and dispersed. Because nonpoint nutrient loadings enter water bodies over a dispersed area rather than an identifiable point, direct measurement of the amount of nitrogen and phosphorous discharged to a receiving body of water is difficult and costly to measure with certainty. Direct market-based policies (a charge on nitrogen and phosphorous discharge for instance) may not be well adapted to address nonpoint source pollution since these policies would require that nonpoint nitrogen and phosphorous discharge be measured, quantified and traced to a specific origin. Indirect incentive-based policies, however, can be applied to the use of the input that causes the pollution problem. A fertilizer tax (charge) on nitrogen and phosphorous fertilizer, for instance, would raise the price of the input that contributes to the nonpoint nutrient problem. A fertilizer tax is currently applied to commercial agriculture in several states and countries (Anderson, Hofmann, and Rusin 1990; OECD 1991, 1993).

The fertilizer tax can be used in two ways to achieve improvements in water quality.

First, the tax could be set so as to substantially reduce the purchases (demand) of fertilizer. Not only will the higher price reduce fertilizer use but the higher input costs also provide an incentive for farmers and homeowners to more closely match fertilizer with plant requirements. The reduced use of fertilizer and more efficient use of fertilizer both function to reduce the total amount of nitrogen and phosphorous leaving the field. Second, a fertilizer tax generates revenues that could be used to fund other water quality programs. Revenues from the fertilizer tax could then be earmarked for related efforts (education and cost-share) to reduce nutrient reduction. Even a small urban fertilizer charge would generate significant revenues. For instance, Maryland estimates that a 2 percent surcharge on the nutrient content in lawn and garden fertilizer would generate between $1 and $3 million annually in state revenues (*Financing Alternatives for Maryland's Tributary Strategies* 1995).

Another advantage of a fertilizer tax is that it would be simple to implement and enforce. The tax could be easily incorporated within the existing sales tax structure. The availability of substitutes for commercial fertilizer, however, could reduce the environmental effectiveness of a fertilizer tax. For instance, as the price of fertilizer increased, farmers may increase manure applications to offset the reduced use of fertilizer. Thus, a fertilizer tax will not always result in a total reduction in nutrient loadings. Therefore, a total nutrient plan is needed.

Pesticide Tax: One market-based policy that could be used to reduce pesticide use is a pesticide tax. While there are a variety of fees currently in place related to pesticides, these do little to alter the incentives people face regarding pesticide use. For instance, many states require pesticide dealers to pay a license fee to operate. Virginia's Pesticide Control Act (1989) also requires those who sell pesticides to pay an annual registration fee for each brand or grade of pesticide offered for sale or use in the State. These types of registration fees do little to change total pesticide use since the charge is not levied on a per unit basis. While designed as a revenue generating mechanism, these fee systems could be modified to provide incentives to reduce pesticide use.

A per unit pesticide tax could be implemented in a number of ways (Williams 1992). One of the most straight-forward applications would apply a charge to all pesticides purchased. This approach is effectively a higher sales tax for pesticides and such taxes have been implemented in several European counties (OECD 1993). Furthermore, the size of the charge could varied between different pesticides based on their toxicity and persistence in the environment. While imposing charges based on the potential environmental risks of a pesticide would complicate the design and implementation of a pesticide tax, such an approach would avoid reducing the use of environmentally benign, but effective pesticides.

Increasing the cost of using pesticides would reduce overall pesticide use. By making pesticides more expensive to use, particularly for the most potentially environmentally harmful ones, a pesticide tax system would accelerate the use of integrated pest management (IPM) practices. Earmarking revenues for IPM research and education would also help alleviate concerns about the distributional fairness of such a tax.

In designing a pesticide tax, care must be taken to institute a system of charges broad enough to cover all potentially hazardous pesticides. If the tax is applied to a narrow range of pesticides the potential exists for the creation of incentives to adopt potentially more harmful substitutes not subject to the charge. If the tax covers a broad range of pesticides, the substitute products and pest control techniques would work as an advantage, not a disadvantage, in reducing use of potentially harmful pesticides.

Deposit-Refund Systems: Many of the most difficult toxic problems to address involves devising policies which ensure the safe use and disposal of hazardous and toxic chemicals from a large number of relatively small users. Two of types problems that fall into this category: the disposal of hazardous household waste and disposal of unused commercial pesticides and pesticide containers. These wastes can present a potential source of chemical contamination of Bay waters when disposed of improperly (Chesapeake Bay Program 1994).

Several local governments in the Bay area have sponsored collection and disposal programs for unwanted household hazardous wastes and agricultural pesticides (Chesapeake Bay Program 1994). The typical program involves designated hazardous waste collection days. Although these types of program has been successful in safely disposing of an existing stock of hazardous wastes and waste containers, they do not alter the incentives people face to properly dispose of their hazardous wastes. Since the disposal of hazardous wastes involves a commitment of personal time and resources, there exist real financial incentives for easy, inexpensive, but improper disposal. While there may exist laws regarding the improper use and disposal of hazardous materials, the large number of diffuse users makes monitoring of safe disposal very difficult. Furthermore, people may not be aware of waste disposal facilities or may no be aware of how to properly dispose of wastes.

A deposit-refund system is a market-based policy that would help ensure that the container and/or unused portion of a hazardous chemical are disposed of properly (Russell 1988; Stavins, *et. al.* 1991). Deposit-refund systems are most appropriate in relatively small volume situations where discharge and disposal is widely dispersed and monitoring difficult (Russell 1988; Stavins, *et. al.* 1991). A deposit-refund system for bulk pesticide containers, for instance, could require a substantial deposit for all types of containers of a certain size. All or a portion of the deposit is refunded when the container is returned to a designated collection site. Any unused contents could be accepted without penalty. The initial deposit and subsequent refund should be set large enough to make the act of returning the container worthwhile (Russell 1988). A deposit-refund system can be described as a self-enforcing system since there exists little incentive for illegal dumping. Since 1985, Maine has operated a deposit-refund system for limited use and restricted use pesticide containers (Anderson, Hofmann, and Rusin 1990).

A deposit-refund system can be designed in several ways to accomplish other objectives. A small difference between the initial deposit and a full refund could be established to pay for the disposal and administrative costs of the program. In this case those who use potential chemical contaminants would also be the ones paying the costs of disposal instead of the general public. The program would be self-financing and would not have to rely on sources of revenue to operate the program. A deposit-refund system could also be partly based on some combination of the toxicity and quantity of the pesticide returned. If a deposit-refund system is at least partly based on the amount of the unused pesticide returned, then an added incentive to reduce amount of pesticide used is created. Such a variation, however, would be more difficult to implement than one where refunds are based only on the container returned. A deposit-refund system operates best if the refunded product is easy to identify and difficult to counterfeit. The large number of available pesticides and the corresponding problems of identification present the opportunity of diluting the returned portion of the chemical, thereby undermining the integrity of the system (Macauley and Palmer 1992).

Improving Agricultural BMP Cost-Share Programs: Unlike the control of point source of nutrient pollution, the primary approach in the Chesapeake Bay watershed for the control of agricultural nonpoint source pollution has been a combination of educational and cost-share (subsidy) programs. Without assigning pollution control responsibilities, this approach relies on the use of scientific knowledge and financial assistance to persuade farm operators to adopt techniques that reduce the amount of nutrients leached into surface and ground water. As a method of persuasion to reduce nutrient pollution, however, the cost-share program can be improved by applying the lessons from market-based policy approaches.

Under current cost-share programs, farm operators are provided cash assistance to implement best management practices that will reduce erosion, pesticide use, and nutrient pollution. While there are a variety of state and federal cost-share programs, they all share some common characteristics. Cost-share money is usually distributed to applicants on a first come basis or shared equally among all who apply for a listed practice. In addition the amount of cost share is typically not based on the full costs to the farm operator for adopting the BMP. Thus, the availability of cost-share assistance does not provide farmers with clear incentives to implement the least cost and the most effective nutrient reduction practices (Dunn and Shortle 1988). In other words there are ways to improve the agricultural cost-share program to provide more nutrient reduction bang per dollar spent.

Market Development

Although often taken for granted, markets are a product of social construction. Surrounding every market there exists a system of rules and norms that structures what buyers and sellers can or cannot and must or must not do. Consider an ordinary trip to the local supermarket. Every time a consumer enters a grocery store, all transactions are ultimately structured by a variety of product liability laws, an approved system of weights, food inspection requirements, product information disclosure requirements, legal safeguards against unfair pricing, and legal remedies for theft. While buyers and sellers freely interact within this institutional structure, the market operates in a socially desirable fashion only when these rules and norms of market behavior are followed by participants. Many people think of markets as ways of allocating private goods like apples, oranges, automobiles and computers, yet markets can be constructed to coordinate diffuse individual actions into achieving society's environmental goals and objectives.

Tradable Permits and Quotas

One of the most discussed market-based environmental policies involves the creation of a tradable permit system (also referred to as tradable allowances or quotas). A system of tradable permits starts by defining a tradable product for the environmental problem of interest. In the case of air pollution, for instance, the tradable product is a permit that allows the holder to discharge a certain amount of emissions within a certain time period. Since the permit does not specify how to control emissions, the permit holder is granted flexibility in meeting the terms of the permit.

Environmental goals are achieved by fixing the total number of permits issued. Unlike the use of effluent or emission taxes that relies on price changes to induce the necessary reduction in pollution levels, a system of transferable pollution permits sets the level of pollution directly by limiting the total number of permits issued.[2] The limitation on the total number of permits issued is sometimes referred to as a "bubble" because the maximum amount of allowable emissions is fixed over the relevant geographical region.

Once the overall number of permits has been established, dischargers are allowed to buy and sell permits. One of the most important distinctions between a tradable permit system and conventional pollution control approach used in the United States is the opportunity to trade pollution control requirements (permits) between dischargers. Under conventional air and water pollution regulations, technologically determined and uniform effluent limitations are written into the individual discharge permits (Ackerman and Stewart 1988) Yet, this approach is costly since similar pollution control requirements are imposed on discharge sources that have different pollution control costs. A system of tradable pollution permits would allow pollution sources with high control costs to contract with low control cost sources to achieve their necessary pollution reduction obligations. High control cost sources get credit for the corresponding pollution reduction and low control cost sources are fully compensated for accepting additional pollution control responsiblities. Through this voluntary and mutually beneficial exchange relationship, the overall target emission levels remain the same, but the least-cost ways of reducing pollution tend to be implemented first.

The opportunity for trade not only lowers the cost of pollution control compliance, but also establishes a constant incentive to prevent pollution discharges in the first instance -- an objective of pollution prevention. In a competitive market a price for pollution permits emerges between the interaction of buyers and sellers. The positive price of the permits creates a continuous and positive incentive for all dischargers to continue to develop pollution reducing techniques and strategies. Since the polluter's effluent discharge cannot exceed their total permit holdings, a permit holder wishing to discharge more pollutants into the environment must purchase additional permits from another holder. Simply holding a permit should not be considered a costless undertaking either, since the positive value of the permits represents a potential source of

[2] The creation of a system of tradable discharge permits has been called a quantity-based incentive system since the regulatory authority controls the amount of pollution released, but not the expenditures for controlling incremental units. In contrast, charge systems fix a tax or subsidy that will be made per unit but leave uncertain the resulting level of environmental quality (Anderson, Hofmann and Rusin 1990).

revenue that must be foregone in order to discharge pollutants. Within a competitive permit market, the financial incentive of dischargers to reduce costs drive the search for more cost-effective pollution control strategies.

A system of tradable pollution permits is also better able to control the overall level of emissions or effluents than the conventional permitting system. While the conventional permitting system limits pollution levels from each discharger and imposes more restrictive discharge limitations on new pollution sources, there is no guarantee that overall levels of pollution will meet environmental objectives. Even if new sources comply with more restrictive permitting requirements, there is no mechanism to ensure that the increase in emissions from new pollution sources will not be offset elsewhere.

Market Facilitation

A market can also be facilitated for an *existing* good or service that promotes environmentally friendly products or outcomes. This class of market policies is labeled here as "market facilitation." There are nearly limitless ways in which markets can be developed and facilitated. Since improved environmental quality is a desirable objective of many people, it would be logical to conclude that people would be willing to pay something extra for products that are produced with minimum environmental harm. The demand for this type of products could be enhanced by providing additional information to the consumer about some environmental aspect of the product. Some retail supermarkets and drug stores in the United States label shelves where product with packaging which meets a standard for environmental compliance can be found (Grogan and Schwartz 1991). Labeling can be used to help manage pesticide toxicity and leachability.

Product development could also involve efforts at providing quality assurances. Currently no clear criteria exist for defining what is meant by organically grown produce. If well-defined, credible, and clearly marked organic content labels similar to the nutrition labeling requirements could be developed for fruits and vegetables, the demand for reduced pesticide or pesticide free produce could increase.

The development of environmentally beneficial markets could be hobbled by trading obstacles such as uncertainty, information problems, and high trading costs. Particularly applicable to new and developing markets, potential traders may be unaware of other interested parties or face high information and search costs in searching for trading partners. The development or facilitation of organizations or mechanisms to transmit this information and reduce search costs can improve the functioning of these markets

Possible Applications of Environmental Markets

Facilitating Manure Markets: Several areas in the Chesapeake Bay watershed contain high concentrations of poultry and livestock operations. Traditionally, this manure has been disposed of on surrounding crop and pasture land. Some of these same areas, however are also experiencing a rapid growth in poultry and livestock numbers, raising the concern that manure production has exceeded the potential for surrounding agricultural land to utilize the manure's nutrient content. In some portions of the Susquehanna River basin in Pennsylvania and the Shenandoah River basin in Virginia, one of the most significant threats to water quality stem from the high concentrations of manure application on land.

Manure can provide an inexpensive source of nitrogen and phosphorous for crop production. Poultry litter can also be used as a supplemental feed source for cattle. The positive economic value of the manure creates the opportunity to develop a type of waste exchange. If operators of intensive livestock and poultry facilities are able to locate and negotiate manure trades with cropland farmers, a marketing opportunity to ship manure from surplus areas to manure deficit areas is created (Bosch and Napit 1991, 1992). In such cases, the potential exists for both parties to improve their profit position while reducing the threat of nutrient pollution. In several locations within the Chesapeake Bay watershed, manure markets and trades are starting to develop (Logdson 1992). This experience, as well as the experience with manure markets elsewhere in the U.S., provides a variety of examples of how manure markets can be facilitated or improved to advance water quality objectives.

In many cases the manure markets have developed by the actions of third-parties who incurred the initial costs of establishing trading relationships. For example, an agricultural Extension Agent in Lancaster County, Pennsylvania conducted a survey to locate potential buyers and sellers of manure. With this information, the agent facilitated manure exchanges by putting buyers and sellers in relatively close proximity in contact with each other (Logsdon 1992). While these third parties incur the initial information costs with establishing a manure market, the need of these informational services may decrease as long-term trading relationships develop. As markets become established, manure brokers and manure handling firms may enter the market to coordinate and conduct manure trades.

The reduction of uncertainties will also tend to facilitate trade of nitrogen and phosphorous from manure surplus areas to manure deficit areas. One uncertainty is that many farmers do not view manure as a valuable fertilizer source (Logsdon 1992, Bosch, Pease, Batie, and Shanholtz 1992). Another potential barrier to the development of manure market is the uncertainty about nutrient content of the manure. Farmers may be unwilling to pay for a load of manure if the nitrogen, phosphorous, and potassium content is variable and uncertain.

The impetus to trade can be facilitated by assigning those who create waste the responsibility for environmentally safe disposal. The assignment and enforcement of this responsibility provides the public with assurances that manure will be shipped from surplus to deficit areas. As the density of poultry or livestock facilities increases, the distance to manure deficit areas grows. It is possible that poultry and livestock production could become so concentrated in a particular area that the cost of hauling surplus manure to deficit areas could exceed the price manure users would be willing to pay for the manure. If animal growers are assigned the responsibility for safe manure disposal, they would then have to incur some of the costs of shipping manure in this situation. By incorporating the consequences of waste generation into everyday business decisions, the explicit assignment of waste management responsibilities may also influence the growth of the poultry and livestock industry in a more environmentally friendly spatial pattern.

Nutrient Trading Systems: The use of a permit or quota system has also been proposed as a way to address the nutrient pollution problem. In several locations across the U.S. nutrient trading systems have been implemented as a means to control nitrogen and phosphorous related water pollution. For a discussion of this type of market-based trading system, a companion publication titled "Market-Based Strategies & Nutrient Trading: What You Need to Know" is available from the same authors.

Liability Systems

Although often overlooked as an example of a market-based environmental policy approach, the establishment of liability rules constitutes an important addition to the set of environmental policy tools. Liability rules establish rights to claim damages for certain injuries to a resource or an individual. Unlike property rules which grants rights of use and control over a resource, liability rules are an *ex post* legal remedy - that is an individual can only seek compensation after environmental harm has been inflicted (Calabresi and Melamed 1972; Landes and Posner 1987).

How does *ex post* compensation translate into pollution prevention and the reduction in environmental risks? More specifically, how can liability rules be considered a market-based environmental policy? If properly designed, liability rules assign financial responsibility of pollution damages to the polluter who caused the harm (Tietenberg 1989; Huber 1988). The possibility of paying for pollution damages creates an incentive for the polluter to seek ways to minimize environmental pollution and risks in a cost-effective way. Systems of liability offers other features absent in fee-based or market-development policies. Unlike other market-based approaches, pollution deterrence is linked directly to victim compensation (Katzman 1986). Thus, liability rules are the only class of market-based policies that explicitly compensate those who have suffered from environmental damages (Tietenberg 1989).

In general there are two types of liability rules: negligence and strict liability. Under negligence, the polluter is required to compensate victims for losses stemming from the failure of the polluter to meet some given standard. Thus, the court or administrative system must determine whether the potential injurer was negligent in efforts to prevent the environmental damage before compensation is awarded. Strict liability, on the other hand, a polluter must pay compensation to victims of environmental damage or accidents *regardless* of the steps taken by polluter to prevent the damage (Calfee and Winston 1988). Compensatory damages are awarded only upon some proof that accused injurer's activities caused or contributed to the damage.

While conceptually straight forward, establishing a system of liability rules can be complicated to implement. A number of difficult issues must be recognized and dealt with in order for liability rules to create clear and effective economic incentives to reduce environmental risks. Under a liability system, the potentially liable party or parties also need to be identified. In the environmental arena, this can be a particularly difficult issue. In the case of spill and other environmental accidents, the identification of liable parties is usually straight-forward. While the magnitude of the damages may be less than clear and hotly contested, the party responsible for the environmental accident is usually easily identified.

Many types of environmental problems, however, are not easily traced back to an identifiable source or party. For example, cases involving diffuse, mass-exposure and latent risks are particularly difficult to trace back to a single liable party. Without a singe identifiable party, victims may have trouble collecting compensation. In these cases, "joint and several liability" could be applied. The joint and several liability rule would make a party who was partially responsible for environmental damage liable for an amount up to the entire damage caused. Thus, if it is impossible to determine the exact individual contribution from a group of polluters that caused the damage, then this liability rule states it is better to have someone pay for damages than no one pay. While joint and several liability provides a mechanism to recover costs for environmental damages, it also presents the potential to unfairly punish parties that have played only small role in the overall environmental harm (Plater, Abrams, Goldfarb 1992).

SUMMARY

Conventional ways of obtaining improved environmental quality are increasingly costly. The market-based environmental policies reviewed above offer many opportunities to improve the environment *and* control the costs of compliance. Furthermore, as the limitations of conventional environmental regulation become more visible, there seems to be increasing political willingness to implement market-based environmental strategies. In many cases agreement between environmental groups and private industry in the use of these policies is offering a unique opportunity to advance environmental goals by building a consensus between these traditionally antagonist groups (Hahn and Stavins 1991).

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