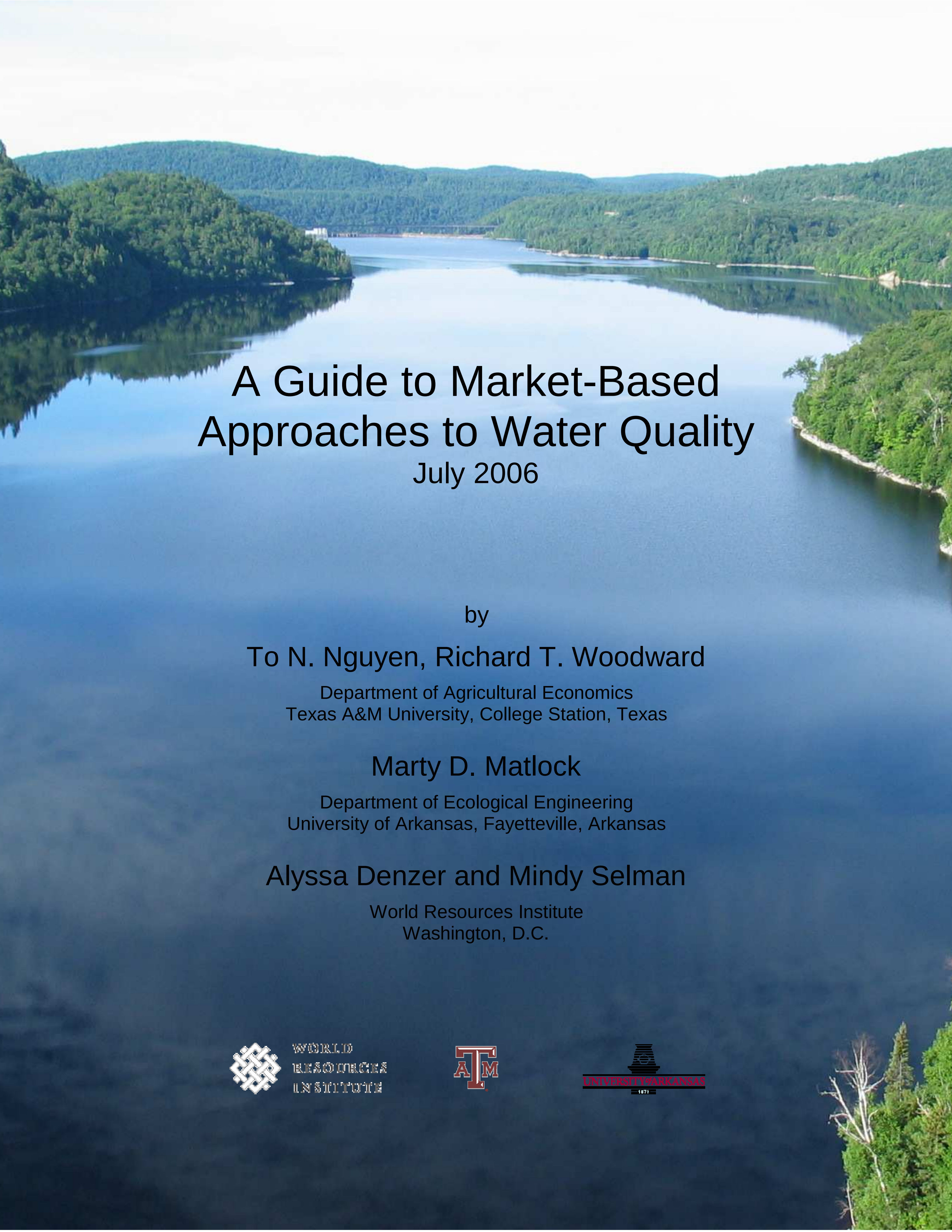

A Guide to Market-Based Approaches to Water Quality

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by

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CHAPTER 1 INTRODUCTION

This guide is intended as an introduction to the problem of water quality and the use of market-based instruments to address the problem. Water quality problems remain a significant concern in the United States and many other areas of the world. Although many of the most egregious problems have been largely eliminated, a substantial share of the nation's rivers and lakes are not able to sustain appropriate habitats and are both unsafe and unsightly. Traditional management approaches are proving insufficient to clean up U.S. waterways. As a result, policy makers are increasingly looking for new approaches to achieve water quality goals. Market-based approaches are one prominent tool under consideration.

This guide consists of four chapters, including this brief introduction. Chapter 2 gives an overview of the problem of water quality, discusses how water quality problems have historically been managed in the U.S. and introduces the available policy options. Chapter 3 focuses specifically on market-based approaches to water quality management. We introduce the basic economic idea, give some examples of how market-based approaches have been used, and discuss the policy environment in which such approaches are being developed. Finally, Chapter 4 discusses how water quality processes or non-point sources are modeled and the inherent uncertainty that results in such models.

This guide is a companion to the site http://edu.NutrientNet.org, an on-line environment in which classes can experience and learn about water quality trading. That Internet site, based on the World Resources Institute's NutrientNet.org, can help students explore the potential for water quality trading.

CHAPTER 2 AN OVERVIEW OF WATER QUALITY PROBLEMS AND POLICIES

Water quality is an ongoing concern in the United States and throughout the world. In this section, we provide a synopsis of the relationship between pollutant discharge and water quality, sources of pollutants, and methods of measuring water quality. We also present the current water quality conditions in the U.S., as well as a brief history and discussion of policies designed to address water quality.

Nutrient Polllution

The cause-effect relationship between pollutant discharge and pollution damages is shown in Figure 2.1. Pollution enters into receiving waters (such as rivers, lakes, and groundwater) through pollutant discharge sites. Receiving waters are able to assimilate only a certain amount of the discharged pollutants depending on their *absorptive capacity*. If the amount of pollution entering the receiving waters exceeds its absorptive capacity, there will be an accumulation of excess pollutants. These excess pollutants can cause damages to the environment, affecting people and the ecological system.

The absorptive capacity of receiving waters differs between pollutants, affecting how easily they can be assimilated. *Stock pollutants* are assimilated slowly. Examples of stock pollutants include heavy metals, persistent synthetic chemicals, and non-biodegradable bottles. Pollutants that are assimilated quickly are called *fund pollutants*. Wastes containing organic matter, which are broken down by the bacteria in water into less harmful inorganic components, is an example of a fund pollutant.

Although both organic and inorganic pollutants contribute to the degradation of water quality, this guide will focus only on organic pollutants. In the case of organic pollutants, water quality declines because aerobic bacteria attack the organic matter and consume large amounts of the available oxygen in the surrounding water. This process leads to one of the most serious side effects of pollution from organic pollutants – reduced dissolved oxygen levels. Related to the oxygen depletion effect, there are two commonly used measures to gauge the water quality: Dissolved Oxygen (DO) and Biological Oxygen Demand (BOD).

Figure 2.1: **Water pollution process**

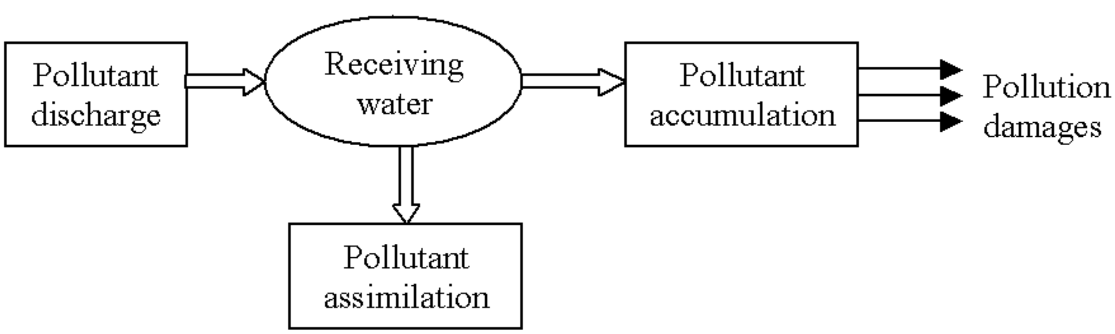


Table 2.1 DO and effects on aquatic life	
DO (mg/L)	**Aquatic Life**
> 4.8	Supported
2.3 – 4.8	Dependent
< 2.3	Not supported
Source: USEPA, 2004a	

Table 2.2 BOD and water quality	
BOD (mg/L)	**Water Quality**
< 2.0	Very good
3.0 – 9.0	Somewhat polluted
> 10	Very polluted
Source: Stevens Institute of Technology, 2004	

Dissolved Oxygen (DO) refers to the oxygen level currently in water, usually expressed in parts per million (ppm) or milligrams per liter (mg/L). Table 2.1 shows some critical oxygen levels and their effects on aquatic life. When microbes such as aerobic bacteria consume oxygen such that DO drops below a critical threshold, other organisms such as benthic macroinvertebrates and fish may die. The duration and frequency that DO levels fall below 4.8 mg/L governs the ability of a water body to support most aquatic life.

Biological Oxygen Demand (BOD) refers to the amount of dissolved oxygen that would be consumed by aerobic bacteria in a water sample over a specified period of time (usually 5 days). Organic matter serves as food for aerobic bacteria; the more organic matter metabolized, the more oxygen consumed. As organic matter in the water increases, the demand for dissolved oxygen also increases. In general, DO levels in streams tend to decline with increasing BOD. Table 2.2 presents a rough assessment of water quality based on BOD levels.

Sources of Nutrient Pollution

Water pollution can be classified into two general categories: *point source* and *non-point source*. Point source pollution includes discharge into receiving water through identifiable and monitorable sources such as pipes, ditches, and outfalls. Municipal and industrial waste discharges fall into this category.

In contrast, non-point source pollution enters the receiving waters through flow paths that are unidentifiable or difficult to monitor. Examples of non-point sources of pollution include agriculture (cropland, pasture, etc.) and residential lawns, which contribute to sediment, nutrient (nitrogen and phosphorus), and pesticide losses to local waterways. Compared to point sources, non-point sources are more difficult to manage through a regulatory process. Generally speaking, regulatory agencies in the U.S. have directed the majority of enforcement dollars and effort at managing pollution from point sources over the past 25 years. Despite the fact that non-point source pollution is now the largest contributor of pollution in U.S. waterways, this form of pollution has been mainly addressed on a voluntary basis.

U.S. Water Quality

The principal source of information about U.S. water quality is the U.S. Environmental Protection Agency's National Water Quality Inventory Report to Congress (305(b) report). This is a compilation of information submitted by states, Indian tribes, territories, interstate water commissions, and the District of Columbia as required by Section 305(b) of the Clean Water Act. In 2000, states and other jurisdictions surveyed 19 percent of U.S. river miles, 43 percent of lakes, ponds, and reservoirs acres, and 36% of its estuaries squared miles (Table 2.3). The survey found:

> For rivers, 53 percent were rated good (fully supporting all designated uses assigned to the waters such as drinking, swimming, or fishing), 8 percent were rated good but threatened for one or more of its designated uses, 39 percent were rated impaired for one or more uses, and less than 0.1 percent for which uses were not attainable.
>
> For lakes, ponds, and reservoirs, 47 percent were rated good, 8 percent were good but threatened, 45 percent were impaired, and uses were not attainable in less than 0.1 percent.
>
> For estuaries, the corresponding percentages are 45 percent good, 3.3 percent good but threatened, 51 percent impaired, and less than 0.1 percent not attainable.

Figure 2.2 shows the status of rivers and lakes as reported in EPA's surface water quality assessments undertaken between 1992 and 2000 (USEPA, 2000). For rivers and lakes the picture is somewhat ambiguous. Of those surveyed, the proportion of rivers that were of good quality improved slightly from 1992 to 1998 but fell to 61 percent in 2000. For lakes, ponds and reservoirs, the improvement between 1992 and 1994 was offset by a decline from 63 percent in 1994 back to 55 percent in 2000. For estuaries, the data reveal serious degradation in water quality with only 49 percent of surveyed waters were of good quality in 2000 compared to 69 percent in 1992.

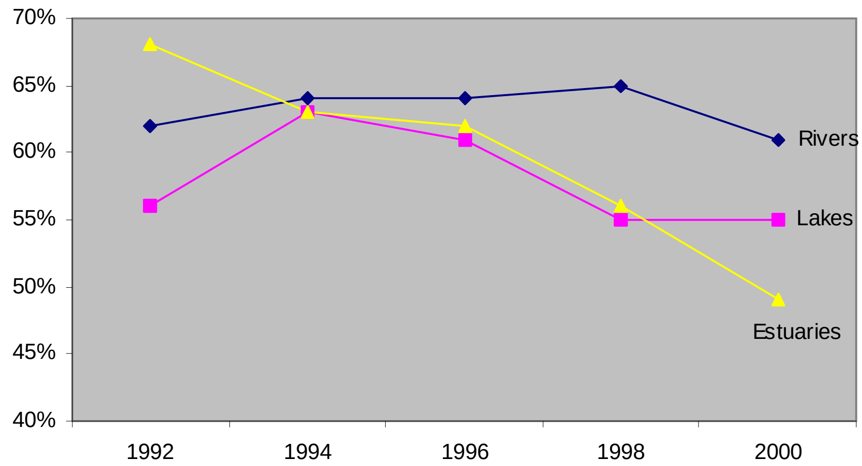
According to The 2000 Water Quality Report (USEPA, 2000), the leading source of water impairment for rivers and lakes in the U.S. was agriculture. Agricultural runoffs contributed to problems in 48 percent of the

Table 2.3: Summary of Quality of Assessed Rivers, Lakes, and Estuaries

Waterbody Type	Total size	Amount assessed (% of Total)	Good (% of Assessed)	Good but threatened (% of Assessed)	Polluted (% of Assessed)
Rivers					
(miles)	3,692,830	699,946	367,129	59,504	269,258
(%)		*19%*	*53%*	*8%*	*39%*
Lakes					
(acres)	40,603,893	17,339,080	8,026,988	1,348,903	7,702,370
(%)		*43%*	*47%*	*8%*	*45%*
Estuaries					
(sq.miles)	87,369	31,072	13,850	1,023	15,676
(%)		*36%*	*45%*	*3.3%*	*51%*

Source: USEPA. *The 2000 Water Quality Report* (USEPA, 2000)

Figure 2.2: **Proportion of good quality waters of assessed rivers and lakes (1992-2000)**



Source: USEPA. *National Water Quality Inventory Report to Congress,* various years.

impaired river miles and 41 percent of the impaired lake acres. Pollutants in municipal discharges are still important, however, degrading aquatic life or interfering with public use in 37 percent of the impaired estuarine waters.

The 2000 report also lists the leading causes (pollutants) of water pollution for each of the three waterbody types. For rivers and streams, the leading causes were pathogens (bacteria) and sediments, for lakes and reservoirs were nutrients, and for estuaries are metals (primarily mercury).

U.S. Water Pollution Policy

Before the late 1940's, local and state authorities took the primary responsibility for water quality policy. Federal influence over water quality began increasing in 1948 with the implementation of the Water Pollution Control Act, which established the legal authority for Federal regulation of water quality. Amendments in 1956 broadened federal authority to allow for direct federal regulation and enforcement of waste discharge. and to give federal subsidies to local wastewater treatment plants.

Federal authority further increased with the implementation of the Water Quality Act (1965) which contained the first federal stipulations for ambient water quality standards for interstate water courses. These became the basis of interstate water quality standards. The Act also required that states have plans in place to ensure these standards were implemented.

Federal authority over water quality and pollution control significantly increased again in 1972, when Congress passed the Clean Water Act (CWA). The basic objective of the CWA is "to restore and maintain the chemical, physical and biological integrity of the Nation's waters."[1] The CWA sought to have "fishable and swimmable waters" by 1983, "zero discharge of pollutants by 1985, and to eliminate the release of "toxics in toxic amounts." The law made the U.S. Environmental Protection Agency (USEPA) responsible for setting national standards for the discharge of effluents on an industry-by-industry basis, considering both the capabilities of pollution control technologies and the costs of implementation. This legislation made it illegal to discharge pollutants to surface waters without a permit, and focused specifically on point sources. The CWA was extensively amended in 1977 and 1987 to expand USEPA's powers and to address non-point pollution through voluntary programs.

Another important component of the original CWA is that it assigned to states the responsibility to review their waterways for impairment. States then were required to provide a listing of all impaired waterways to the USEPA, and also to establish Total Maximum Daily Loads (TMDL) for the limiting pollutant in each impaired waterway. TMDLs are designed to achieve applicable water quality standards for the impaired waterbody. A water quality standard is a qualitative standard of uses such as swimming or fishing that a water body must be able to support.

A complete TMDL analysis includes two major components. The first is the calculation of the maximum pollutant amount that a water body can receive and still meet the water quality standards. The second component allocates the total maximum daily load between point and non-point sources. In addition, the calculation of the load must provide for a margin of safety and seasonal variability in order to ensure water quality standards are consistently being met. In practice, however, there are several difficulties in determining and implementing TMDLs.

The first problem facing the implementation of TMDLs is the technical challenge of determining a single allowable daily load. Changing meteorological and climatological conditions can cause the true maximum load to vary dramatically over the course of the year. In addition to this technical problem, TMDL regulations have been extraordinarily controversial. On the environmental side, there are legitimate claims that the water quality goals set forth in the CWA have not been achieved, and, as noted above, it appears that the U.S. may even be backsliding (see Figure 2.2). Cities and businesses, on the other hand, often object to TMDL implementation because the costs of meeting new water quality goals can be quite high.

Since the Water Pollution Control Act of 1948, water pollution control policy in the U.S. has mainly relied on the regulatory approach, employing laws and regulations to mandate uniform standards for polluters. Even though it has been quite effective in reducing point-source water pollution, the regulatory approach has left largely ignored the non-point source pollution problem. Aside from programs such as the Conservation Reserve Program and the Environmental Quality Incentives Program (EQIP), which provide incentives to farmers to reduce erosion and control nutrient loads, the traditional federal regulatory policy has focused on point source pollution. Non-point source pollution has been considered the responsibility of the states.

Choosing the appropriate policy instrument to improve water quality is especially difficult with respect to non-point sources. First, non-point source pollution involves diffuse sources, meaning it is difficult to identify the exact point/location that the pollution is entering the waterway. Second, random events such as weather can effect the runoff load. These two factors make it costly or even infeasible to efficiently and effectively measure and monitor non-point source pollution. Third, site-specific factors may also affect loading and an efficient policy must take these factors into account. However, gathering and analyzing information on every site can be quite costly.

Because of the difficulties with monitoring, mandatory standards for non-point source pollution loads are difficult, if not impossible, to apply. Instead, regulations are typically applied to input factors. For example, with agriculture regulations may be placed on fertilizer use or allowable agricultural practices. Such regulations are often based on models that predict pollution loads from farms and the overall effect of farm management choices on water bodies.

A second problem with the traditional regulatory approach is that it is typically not cost effective. That is, the environmental results that the policies' achieve are more costly than is necessary. By mandating universal standards for pollution discharge levels or technology for point sources, the regulatory approach may generate two sources of cost-ineffectiveness: one among the point sources, and the other between point sources and non-point sources.

First, the combination of mandating pollution discharge standards and restricting technology choices reduces point sources' flexibility when it comes to finding low cost solutions. As we will discuss in more detail in the next chapter, a policy can only reach an environmental goal at least cost when pollution sources have the same unit cost for the last unit of pollution reduction. Because a universal discharge standard sets maximum levels that polluters can discharge, regardless of their abatement cost, the policy may forego cost-savings.

Second, in many cases the conventional regulatory policy has now exhausted the 'low hanging fruit' – the low cost pollution abatement opportunities for point sources are no longer available. Further discharge reductions will require more stringent standards and, therefore, have higher costs. This widens the abatement cost gap between point source and non-point source polluters, with an increasing loss of cost-effectiveness. Furthermore, as non-point source pollution now accounts for the majority of the remaining water quality problem, the most cost-effective approach lies not with further regulation of point sources, but in the reduction of pollution from non-point sources, primarily agricultural sources.

After decades of improving water quality through the regulatory approach, this progress now seems to have leveled off. Any further improvement of water quality in the U.S. requires policy instruments that involve non-point sources of pollution. The next chapter is dedicated to looking at a policy tool that offers the potential to achieve a least cost solution to both point and non-point sources of pollution.

Endnotes

[1] Adapted from Adler et al., 1993, and Arbuckle et al., 1993.

CHAPTER 3 MARKET-BASED ENVIRONMENTAL POLICIES

The market-based approach in which tradeable emission permits are used has grown in popularity in recent years as a way to address water quality problems. In this section, we discuss this option in more depth, detailing the economic principles behind market-based approaches and discuss some of the real obstacles that must be overcome before it can be used successfully.

The Economics of Pollution Control

In economics, water pollution is called a *negative externality* because the costs of the pollution are not borne by the polluters themselves, *i.e.*, the costs are *external* to the polluter. Because the costs are external, producers such as firms or farmers do not take full account of these costs when making choices and usually create more pollution than would be socially optimal. A key role of environmental policy is to resolve this problem. In so doing, two key questions must be answered: by how much should pollution be reduced (the goal), and what policy should be used to achieve this goal (the instrument). Economics provides two criteria to answer these questions: *economic efficiency* and *cost-effectiveness*.

Economic Efficiency

A level of reduction in water pollution is economically efficient if no other level will yield greater net social benefits (benefits less costs). Reducing pollution has positive impacts on public health, habitat health, recreation, and other economic activities, yielding benefits to society at large. On the other hand, reducing pollution costs money. A reduction in water pollution, therefore, is said to be efficient if the benefits to society outweigh the costs imposed on the polluters. If pollution is at its socially efficient level, then the cost of any further reduction would be greater than the benefits.

Although it is easy to define the idea of economic efficiency in theory, it is very difficult and/or extremely costly to quantify all the benefits and costs of pollution reductions. So, in practice, policy makers rarely seek to exactly achieve the economically efficient level of water pollution. Instead they focus on rough benefit-cost tests, rules of thumb, qualitative water quality standards, and/or general guidelines such as safety standards for human or ecological health.

Cost-Effectiveness

The question of instrument choice and the economic criterion of cost-effectiveness have a great deal of applied relevance. Cost effectiveness is concerned with the allocation of pollution reduction responsibilities, taking into account differences between sources. In a cost efficient policy, a source should reduce pollution only if it can do so at a lower cost than other sources.

This principle is illustrated in the stylized example in Table 3.1. Assume there are two discharge sources for a given pollutant, Firm A and Firm B, into a lake. Also, assume the pollution control authority seeks to reduce pollution by 3 units. The cost of reducing pollution by these two sources is given in the table. For example, for Firm B, the cost of the first unit of pollution reduction is \$1 and the cost of the second unit is \$2, so that the total cost of reducing 2 units would be \$3.

A cost comparison on a unit-by-unit pollution reduction basis is outlined in Table 3.2. In option 1, Firm A spends \$4 for the 3rd unit of reduction, while the first unit of reduction by

Table 3.1 Example of a two-firm pollution reduction problem

i^{th} unit of reduction	**Firm A** (Additional cost)	**Firm B** Additional cost)
1^{st}	$2	$1
2^{nd}	$3	$2
3^{rd}	$4	$4

Firm B costs only $1—the total cost of the policy falls if Firm A's abatement is decreased by one unit and Firm B's abatement is increased by 1 unit (option 2). Similarly, with option 4, Firm B spends $4 for the 3rd unit of reduction, while the 1st unit of reduction for Firm A is $2. Therefore, it would be cheaper for Firm B to reduce 2 units and Firm A to reduce 1 unit of pollution (option 3). Ultimately, option 3 is the most cost effective. In this option, each firm undertakes the amount of reduction up to the point that it can do so at lower cost than the other.

More generally, cost effective pollution policies are those with an outcome in which the cost of the last unit reduced is about equal across all sources. The development of policies that lead to such an outcome has been one of the primary interests in environmental economics.

Comparing Basic Approaches to Environmental Policy

Based on the cost-effectiveness criterion , we compare different approaches to water pollution control. These approaches differ in the instruments they employ and the results they obtain.

Regulatory Approaches

The regulatory approach, ofter termed as command and control, uses laws and regulations to mandate in detail the measures such as standards, technologies, and input uses that polluters must adopt under penalty of fines or other sanctions. As discussed in the first chapter, water pollution control for point sources in the U.S. has traditionally relied on this approach.

Regulatory approaches typically are not cost effective for two reasons. First, regulations tend to lock firms into certain practices or technologies. A regulation that requires the installation of a certain kind of filtration

Table 3.2 Possible ways to reduce pollution by three units

Firm	**Option 1** (units)	**Option 2** (units)	**Option 3** (units)	**Option 4** (units)
A	3	2	1	0
B	0	1	2	3
Total cost	$9	$6	$5	$7

system at wastewater treatment plants, or a certain tillage practice on farms does not allow individuals to look for creative ways to reduce their pollution at lower cost.

The second main reason for this lack of cost-effectiveness is that most regulatory policies do not take advantage of the cost heterogeneity across polluters. Using the example in Tables 3.1 and 3.2, we see that it may not be cost effective to have all sources take the same actions. If all firms are required to reduce their pollution by the same amount, then there is no way to allow those that can reduce their pollution at a lower cost to do more. This results in a regulatory approach that, in aggregate, costs the industry more. This not only is bad economically, but can be bad environmentally as well, since the more expensive an environmental policy is, the more likely it is that industry will resist such policies.

Incentive-based Approaches

As an alternative to the regulatory approach, there are a variety of incentive-based approaches that use economic incentives to achieve environmental goals. These approaches allow polluters the flexibility to find the least-cost pollution control option by themselves. The two main types of market-based instruments are *emission charges* and *emission trading*.

Emission Charges

An emission charge (or pollution tax) is a fee imposed on each unit of pollution emitted. Polluters therefore pay an amount equal to the fee times the units of pollution emitted. The tax on emissions gives polluters an incentive to reduce the pollution up to the point where it is cheaper to pay the tax than reduce pollution.

Using the hypothetical case described previously, suppose a fee of $2.50 were charged for every unit of pollution emitted. For Firm A, the first unit of reduction costs $2, which is less than the fee. Hence, it is cheaper for the firm to reduce one unit of pollution than it is to emit that unit and pay the tax. However, Firm A will not want to reduce more than 1 unit because reducing a second or third unit is more expensive than the fee. For Firm B, on the other hand, you can see that the $2.50 fee provides an incentive to reduce 2 units. Hence, this fee leads to the cost effective result, option 3 in Table 3.2.

A challenge with an emission charge is that the fee needs to be set carefully to achieve the desired goal. If the fee in the example were $1.50 per unit of pollution, then Firm A would not reduce anything and Firm B would only abate 1 unit; the goal of 3 units of reduction would not be achieved. Hence, policy makers may have to use a trial and error approach to achieving their desired level of pollutant abatement.

Emission Trading

The second type of incentive-based policy involves a market in which a tradable emission permit is used. In these programs, often referred to as cap-and-trade programs, the pollution control authority sets an aggregate limit and the initial allocation of pollution allowances such that the aggregate cap is not violated. Polluters can then choose to either meet their permitted discharge levels by themselves, or trade. The approach is most easily understood through an example.

Again, consider the case presented in Section 3.1.2. Assume that Firm A and Firm B currently emit 7 and 6 units, respectively, but the pollution control authority is requiring a reduction to 10 units of pollution, 5 by each firm. Under the initial allocation, therefore, the firms would need to reduce 2 units and 1 unit, respectively. This initial allocation is equivalent to Option 2 in Table 3.2, which is not the most cost effective. However, Firm A and Firm B can trade. For example, instead of paying $3 to make its 2nd unit of pollution reduction, Firm A could offer to pay $2.50 to Firm B to purchase one of its allowances. Firm B would accept this offer since $2.50 is more than its $2 cost to make a second unit of reduction. If such a trade takes place, then the firms reach the most cost effective outcome, Option 3.

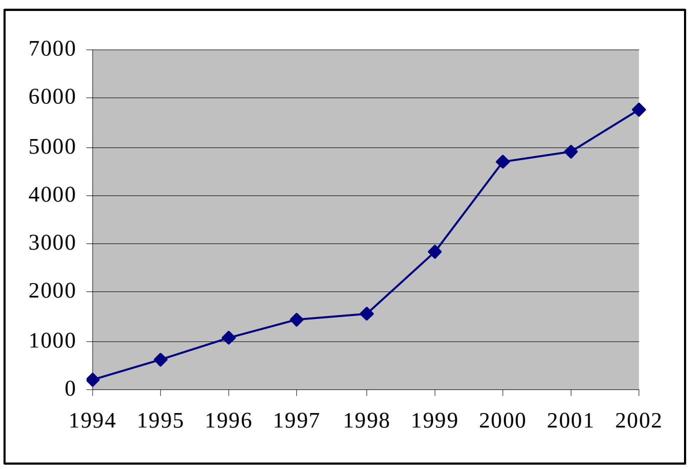
Like an emissions charge, the market-based approach is cost effective. The environmental target is achieved at the lowest possible cost. Compared to an emissions charge, its advantage is that the pollution control authority is assured that the pollution target will be reached without any of the possible adjustments needed with an emission charge. Establishing such a market is not always easy or cheap, but despite these difficulties a market-based approach can often be preferred.

The SO_2 Trading Program

An example of a successful emissions trading program in the U.S. is the Sulfur Dioxide (SO2) Allowance Trading Program. Under Title IV of the Clean Air Act Amendments of 1990, USEPA established the Acid Rain program, including a tradable permit system applied for sulfur dioxide (SO_2), a primary cause of acid rain. The program targeted a 50 percent reduction in nationwide SO_2 emissions from 1980 levels by 2000. The innovative component of this program was its private market which allowed polluters in the program to satisfy their emissions limits either by cutting their emissions or purchasing allowances from others. The attainment of the target was planned in two phases:

> Phase I (1995-2000) covered only the largest, highest emitting electric utility plants. Each year, USEPA allocated each individual plant a number of allowances—each allowance representing a ton of SO_2 emissions.
>
> Phase II (2000 onwards) included virtually all fossil-fuel electric utility plants, even the smaller, newer, and cleaner plants. Furthermore, emissions

Figure 3.1: **Number of transactions in SO2 Trading Market (1994-2002)**



Source: USEPA, 2004c.

limits imposed on the plants covered in Phase I were tightened.

In addition to the annual allocation of allowances, USEPA also auctions approximately 3 percent of total available allowances to provide price information to the market. Penalties are used to encourage compliance; plants that emit more SO_2 than their allowance are charged $2,000 per ton of excess SO_2 emitted.

The SO_2 Allowance Trading program is widely considered a success in achieving its environmental goal at least cost. The environmental target has been reached and exceeded, and the total abatement costs are substantially lower than the non-trading scenario (Stavins, 1998). As shown in Figure 3.1, the total number of transactions in the SO_2 market grew over time, from 215 transactions in 1994 to 5,755 in 2002.

Emissions Trading for Water Quality (Water Quality Trading)

The preceding discussion shows that, in principle, a market-based approach offers significant benefits. It achieves the pollution goal but does so at lower cost to polluters. The underlying driver for trading is the heterogeneity in cost for different polluters to reduce their pollution discharges. In practice, opportunities have been found for trading between *point and non-point* sources as well as between *point* sources.

Non-point source pollution, in particular agricultural runoff, is a major contributor to water pollution and remains largely unregulated. One estimate suggested the cost of non-point source pollution abatement could be 65 times cheaper than that of point sources (Bacon, 1992). Such large cost discrepancies would create great opportunities for trading between point and non-point sources.

Trading between point sources also offers cost-savings and is the most common form of water quality trading to date. Wastewater treatment is characterized by significant economies of scale; large facilities can process wastewater at a much lower per-unit cost than small ones. As a result, the large facilities are likely to over comply with their permitted discharge levels and sell their excess reductions to small facilities. Of course, trading between *non-point* sources is also possible. However, currently no programs exist in which these types of trades are a major component.

How Water Quality Trading Works

In order for trading between polluters to occur, the pollution control authority needs to mandate a limit, called a cap, on the total pollution discharge within an impaired area such as a watershed or a basin. Where a USEPA required TMDL has been completed, this should define the total maximum daily load for each relevant pollutant and establish the initial allocation of the cap to the various polluters in the watershed. Polluters generate credits if they reduce pollutant emissions below their mandated loading level and become potential suppliers of credits. Those who seek to discharge in excess of their allowed loadings are potential buyers for credits. The credits are transferable so that polluters can either buy or sell them. As such, a trading program creates a market for water quality credits and offers polluters the flexibility to either reduce pollution internally or finance comparable reductions that is undertaken by others. Box 1 illustrates how water quality trading works.

Key PracticalIissues forIimplementing Water Quality Trading

When designing a water quality trading program, several issues require special attention. Specifically, attention must be paid to establishing the appropriate baseline for trading, addressing uncertainty, the initial allocation of credits, the monitoring and enforcement of BMP implementation, liability issues, and the structure of the market. This section will address common issues that arise when trying to implement a water quality trading program.

Allocation Issues

As we have shown above, a market-based approach starts out with a decision as to who is responsible for reducing how much

Box 1: An illustration of how water quality trading work (adapted from Faeth, 2000)

Assume there is a river and the state environmental agency determines that the river is impaired. There is too much phosphorus entering into the river, causing algal blooms that die and leave too little oxygen in the water for fish to survive. The state decides to impose a 1mg/L limit on the phosphorous concentration of discharge from municipal and industrial wastewater treatment plants. The state also agrees to let these dischargers have the option of trading to meet this new regulatory requirement. There are three towns along the river and a number of factories. The area also has many farms that contribute to the problem, but these aren't regulated.

For simplicity, also assume each of these plants discharge 10,000 tons of water per day. So, the 1mg/L limit on phosphorous concentration requires them control the phosphorous loading at below 10 tons per day.[*]

The first town's wastewater treatment plant is old and due for an upgrade. That town decides to rebuild its plant and add the necessary technology to meet the standard. The loading of the new plant is 8 tons of phosphorus per day, and meets their new permitted phosphorus limits. When bundled into the upgrade, the additional phosphorus control is cheap, so the town's costs to meet the regulation are low.

The second town rebuilt its wastewater treatment plant five years ago, when the state didn't require phosphorus treatment. Currently, the phosphorous loading from this plant is 15 tons of phosphorus per day and so this plant needs reduce 5 tons of phosphorous loading. Upgrading a relatively new plant is expensive. However, there is factory nearby that also has to meet the new standard. The factory offers to reduce its phosphorus concentration to 0.5 mg/L, sufficient to provide credits to the town to meet it's requirement. Purchasing reduction from the factory is less expensive than upgrading the town's facility, so the town agrees to the trade. Since both the factory and wastewater treatment plant discharge to the river at an identifiable location, i.e., a pipe, this is a point source-point source trade.

The third town also upgraded its wastewater treatment plant recently and the current phosphorous loading is 5 tons above the allowed loading. A member of the town council gets a group of farmers to agree to put conservation practices in place if the town will pay the expense. These conservation practices not only reduce the amount of phosphorus getting into the river, but also sediment and pesticides. The state gives its approval, but the agricultural loads are variable since they only occur when it rains, so the state makes the town buy three times as many reductions as the load reduction requirement for the plant. Since the load reduction requirement of the wastewater treatment plant is 5 tons of phosphorus a year, the town must buy 15 tons of reduction from the farmers. This is a point source-non-point source trade.

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* Here we just use the simplified formula:
Nutrient load (mass) = Concentration (mass / volume) * Flow (volume)

pollution or, the converse, who is initially allowed to emit how much. The distribution of these *allowances* is one of the most important and politically contentious steps in the design of a trading program. For example, suppose a program has 2 polluters and an aggregate cap of 100 units. The cap can be achieved if both polluters are allocated 50 credits each or if one polluter receives 100 credits and the other none. Both allocations achieve the environmental goal—100 units are emitted— but the balanced allocation is almost certainly the more politically acceptable of the two.

The allocation of initial allowances for a cap-and-trade system can be handled in numerous ways, two of which have received the most attention: *grandfathering* and *auctions*. If allowances are grandfathered, emitters receive allowances in proportion to their historical discharge levels. For instance, if a 20 percent reduction in pollution is sought, then all emitters may receive allowances equal to 80 percent of their historical emissions/discharges. Grandfathering is seen as a fair approach, and therefore generally receives greater support from stakeholder groups.

If allowances are auctioned, then emitters are not initially allocated any allowances, but are required to purchase the allowances or tradeable emission permits they need. Implicit in an auctioning approach is the idea that society holds the right and polluters have to pay for that right. Auctions offer the additional advantage that they generate revenue, which can offset the need for taxes. Finally, auctioning has the advantage of giving an immediate price signal to the market, which can facilitate trading. Because of this feature, the U.S. SO_2 trading program has a small auction component combined with an allowance allocation that is based primarily on grandfathering. Because auction impose immediate costs on the industry involved, they do not gain political support from the affected industries and have not been widely adopted as a way to distribute all allowances.

Establishing Baselines

When nonpoint sources are involved in a trading program, the use of clearly defined allowances is usually not possible. In such situations, a strict cap and trade approach is not viable. Instead, policy makers adopt a *baseline and credit* approach in which water quality credits are granted to sources that reduce their loads below a previously established baseline. The setting of the baseline, therefore, is of critical importance both to the environmental integrity of the program and its political viability.

The baseline for non-point sources may be performance-based or practice-based. In agriculture, practice-based baselines usually refer to a minimum set of practices, for instance nutrient management plans, conservation tillage, etc., that are required to be implemented. With these 'baseline' practices in place, the non-point source generates an estimated amount of runoff. To generate water quality credits, which the farmer could sell, the farmer must implement additional conservation measures beyond those required in the baseline. Credits are generated only from these additional measures, not the required ones. On the other hand, performance-based baselines refer to the level of runoff from the farm. Since directly measuring runoff from non-point sources is problematic, tools such as watershed models, field tests, and in-stream monitoring are used to estimate runoff.

Either of these methods will introduce a degree of doubt. Unlike point sources where nutrient discharges are measured at the pipe, non-point sources must rely on broader estimates to establish baseline discharge levels. For a more complete discussion of some current modeling tools and a further discussion on uncertainty that is introduced through modeling, please see Chapter 4, "Modeling Non-point Source Impacts on Water Quality."

A critical feature of any baseline is that it reflects historical practices that were in place prior to the establishment or even the dialogue surrounding the establishment of the trading program. A source must not be able to manipulate its baseline. If this criterion is not satisfied, then sources will have an incentive to increase their loads in order to raise their baseline so that they can then sell those loads later. As obvious as this sounds, it is often difficult to achieve because landowners may lack the records necessary to document their historical practices.

Monitoring and Enforcement

A central tenet of market-based environmental policies is that the tradeable unit must be *real* and *verifiable*. For example, there is a high degree of trust in the SO_2 program because all polluters regulated under this program must install a Continuous Emissions Monitoring System (CEMS) which samples and records emissions over time. These recorded emissions give regulators a direct measurement of SO_2 emissions for each plant making it straightforward to check that all plants have sufficient allowances to cover their emissions. Unfortunately, there are few cases where emissions can be as effectively monitored as is done by the CEMS.

For water quality, regular monitoring of point source discharges is common, and increased monitoring is certainly possible. However, loads from non-point sources cannot be monitored with the same level of precision. As a result, reductions generated by non-point sources are based on the *predicted* load reduction from the implementation of best management practices, rather than *actual* measured runoff reduction. Furthermore, actual runoff and reductions is inherently variable due to weather and other factors.

Uncertainty is also introduced because of imperfect implementation of best management practices. Some monitoring is required to ensure that landowners have implemented their stated management practices. However, even in the best of worlds, it is likely that there will remain some uncertainty about the load reductions achieved by non-point sources.

Uncertainty and Trading Ratios

There is inherent uncertainty in any pollution control program. Even when emissions are accurately measured, the eventual impact on the environment is subject to uncertainty. When pollution loads are not accurately measured or are only predicted using computer models, the uncertainty grows accordingly. Discount factors, usually called trading ratios, are frequently applied to the load reductions that are uncertain. For example, several programs impose a 2:1 trading ratio on reductions that are derived from non-point sources. This means that for every 2 pounds of nutrient reduction that a non-point source generates, only one 1-pound credit is generated. Hence, if a point source wanted to offset a 100 pound increase in its loads, it would have to purchase a predicted 200 pound reduction in nonpoint source loads.

The logic for these trading ratios is that it provides a higher level of certainty that the pollution reduction goal is actually achieved. It should be noted, however, that in some situations it might be desirable to have a trading ratio less than 1:1 since this will have the effect of lowering the price of nonpoint source credits and encourage more participation by such sources.

The trading ratios that are applied to address uncertainty are quite different from those that are used to adjust for the geographical dispersement of pollutants. Because of spatial distribution and hydrologic conditions, it is often the case that emissions at different locations have very different impacts on the environment. For example, pollution reductions by a source right next a degraded lake might have much more impact than reductions far up in the watershed. Trading ratios can easily account for that kind of spatial variability.

Enforcement and Liability

Obviously, enforcement must go hand in hand with monitoring and if violations are found, someone must be liable. Liability might fall on either the seller of the credit (e.g., a farmer producing credits) or the buyer (e.g., a wastewater treatment plant that needs to increase its load. This issue has been one of the biggest barriers to creating a vibrant market for water quality trading (King and Kuch, 2003).

In most point-nonpoint trading programs, it is felt that there would be fewer buyers than sellers, and that buyers would be more easily monitored. Because of this, some have advocated an approach of buyer liability. However, if buyers are held liable, they may be resistant to trading and markets may be slow to develop. For instance, if a buyer is held liable for failed BMPs, especially because of the lack of implementation, there is both little incentive for the farmer to take care in implementing the BMP, and no way for the buyer to estimate the cost of their exposure. This creates additional barriers to trading. Most point source managers have very little knowledge of farming practices, and little ability to verify that BMPs are being implemented properly. Yet the lack of this kind of knowledge and expertise could prove costly if they are liable in a trading program. Finally, with buyer liability a purchased credit is both a right (to emit more pollution) and a responsibility (to monitor the generator's activities). This makes the credits much more less uniform and creates an important barrier to trading.

The alternative, of course, is to use a program of seller liability. In this case buyers of credits are free from the need to monitor farms, but that burden is imposed instead on the government agency or third-party charged with ensuring that environmental goals are met. Buyer liability works in the SO_2 program and the result is that all allowances are equal. However, the challenge of monitoring and enforcement must fall on someone, and in the case of buyer liability it will fall entirely on the government that approves and monitors the allowances. When it is difficult to enforce the requirement that all polluters have sufficient allowances, this creates a risk for government agencies.

One potential option to address liability is to allow third-party agents to act as brokers. Private brokers could bank and sell credits, assuming the liability for assuring that BMPs are implemented. This scenario could be beneficial in several ways: first, it provides a true fiscal incentive for the private broker to ensure compliance among its participating credit sources, helping to alleviate some of the concerns expressed above with regard to third-party monitoring. Second, removing liability from the buyer's side could make point sources, and other potential buyers, more willing to participate in trading, secure that they will not be held responsible for failed BMPs.

Market Structure

Not all markets are created equal. The New York Stock Exchange, for example, is a fluid trading environment in which assets worth millions of dollars can be transacted in seconds at a cost that is trivial in comparison to their value. In contrast, purchasing a used car from a private party may take weeks of searching and researching and involve non-trivial expenses in advertising and government fees.

Similarly, markets for allowances or tradable emission permits also take on a wide range of forms. Allowances in the U.S. SO_2 trading program, for example, are transacted on the Chicago Board of Trade. Such fluid markets have yet to arise for the case of water quality trading, but instead take three predominant forms (Woodward et al., 2002). The first and most prevalent form is *bilateral negotiations*, in which buyers and sellers directly interact to carry out a transaction. The second market form is that of a *clearinghouse*, in which the government or third party broker (e.g., bank) acts as an intermediary between those that generate credits and those that demand credits. Finally, some water quality trading programs allow for *sole-source offsets* in which trading per se does not take place, but emitters are given the opportunity to find alternative ways of reducing their environmental impact on the watershed through, for example, creek restoration.

Some of these market structures are more efficient than others – buying and selling goods on an exchange is much cheaper than doing so via bilateral negotiations. Policy makers should attempt to design trading programs so that the most efficient market possible arises. On the other hand, we should also be aware the most efficient market structure, the exchange, may not be feasible. Just as used cars cannot be sold on the New York Stock Exchange, not all tradable emission permits will be available on the Chicago Board of Trade. While market designers should strive for an efficient market; expectations must be realistic.

The USEPA Water Quality Guidelines

In order to encourage and support states and tribes in developing and implementing water quality trading programs, the U.S. Environmental Protection Agency issued a Water Quality Trading Policy in 2003 (USEPA, 2004d). This policy provides guidance on how trading programs can conform to the Clean Water Act and other regulations that serve as a legal basis for trading.

The guidance from USEPA covers many aspects of water quality trading; some of the most important being:

i) What pollutants should be traded?

Trading of nutrients or sediment loads are encouraged. But for some other pollutants, *e.g.*, selenium, trading programs should be under high scrutiny. In general, the trading of persistent bioaccumulative toxic pollutants is not encouraged by USEPA.

ii) Where should trading take place?

The USEPA states that the trading area, where polluters can trade with each others, should be within a watershed or an area with a TMDL approval. USEPA supports trading not only in watersheds with poor water quality, but also in unimpaired waters, where water quality standards are being attained, as a way to preserve good water quality.

iii) When can pollutants be traded?

USEPA supports trading not only in areas under a TMDL, but also *pre-TMDL trading*. In the latter case, the trading is encouraged to achieve progress towards applicable water quality standards, e.g., Chesapeake Bay. Furthermore, if trading does not attain the standards, a TMDL should be developed.

iv) What should be the baseline for credits?

The baseline is the level of pollutant discharge below which a reduction will generate credits for trading. In the case of trading under a TMDL, load allocation to sources will be the baseline for generating credits. In cases of pre-TMDL trading, the baselines for point-sources and non-point sources should be derived from and consistent with water quality standards or applicable state regulations.

The USEPA's 2003 Trading Policy also discusses how trading programs should take into account a number of relevant provisions of the CWA and implementation regulations. The Policy discusses USEPA's perspective on the compatibility of trading with provisions such as the "anti-backsliding" provision and the requirement for public review of all revisions in NPDES permits.[1] Although overcoming these regulatory hurdles is a challenge, in general the EPA believes that trading can be consistent with the CWA. Finally, the Trading Policy includes guidance for those developing trading program, including a list of general elements for trading programs to be credible and successful.

The publication of the Trading Policy was important in the development of water quality trading in the U.S. because it gave trading a strong vote of support from the institution charged with enforcing the nation's environmental regulations. It may, therefore, prove to be a milestone, clearing the way for much wider use of water quality trading.

Case Studies

Despite the longstanding interest in water quality trading, the number of programs in which there has actually been trading and an active market is quite limited. In this section, we discuss two programs where trading has actually taken place: The Tar-Pamlico watershed trading program in North Carolina, and the program at Lake Dillon, Colorado.

Tar-Pamlico

In 1989, the North Carolina Environmental Management Commission designated the Tar-Pamlico basin as nutrient sensitive water. The formal designation required the state Division of Environmental Management (DEM) to identify the nutrient sources, set nutrient limitations, and develop a nutrient control plan. The DEM analysis showed that most of the basin's nutrient loadings (primarily nitrogen, but also phosphorus) came from agricultural runoff and other non-point sources. Other smaller sources included municipal wastewater treatment plants and industrial and mining operations—all point sources.

The agency originally recommended regulations to halt point source pollution increases. A coalition of point source dischargers then formed the Tar-Pamlico Association, proposing water quality trading as an alternative to control water pollution. The Association's proposal had two important components. First, the aggregate nutrient load of the Association as a whole would be regulated, thus allowing for trading between its members as long as the Association's cap was achieved. Second, members of the Association could pay into a fund that would pursue non-point source reductions to offset any loadings beyond their allowed levels.

As a result of the association's proposal, two types of trading take place in this program. One is point source-point source trading via *bilateral negotiations* between members of the association. The other is point source-non-point source trading via a water quality *clearinghouse* between members of the association and farmers. This clearinghouse, managed by the North Carolina Agricultural Cost Share program, pays farmers 75 percent of the cost of implementing best management practices (BMPs) that reduce the runoff of nitrogen and phosphorus. By 1996, members of the association had purchased $900,000 worth of credits, with $750,000 being contributions to the non-point source fund (Green 1997).

Lake Dillon[2]

In the early 1980s, there was growing concern about water quality in Lake Dillon, located in the mountains of Colorado. In 1982, a study identified the maximum load of phosphorus in the lake and a cap was imposed. Two years later, trading was approved in which the wastewater treatment plants could exceed their limit by reducing the loads from non-point sources, primarily septic systems.

Despite having a trading option, point sources made dramatic reductions in their own loads, far below the 1982 cap and, as a result, the need for trading did not arise for many years. Finally, in 1999 the Copper Ski Area sought to expand its base facilities that would increase its load by approximately 40 pounds per year above its allocated cap; the need for a trade arose.

After reviewing its options, a trade was negotiated between Copper and the town of Frisco, which identified homes willing to switch from their septic systems to the town's sewage system. Earlier studies had found that converting a home from a septic system to a municipal system lead to a net reduction of about 1 pound per home per year. The program's 2:1 trading ratio, therefore, implied that 80 homes would need to be enrolled to satisfy the cap. Frisco found the necessary homes and Copper partially compensated them for the cost of switching to the municipal system. The trade was complete.

There were numerous benefits of this trade. To Copper, the benefit of the trade was substantial; the value to them of these credits was at least $1.5 million. To the town of Frisco, the project increased its client base. For the homes that enrolled, the trade allowed them to get a desirable service at reduced cost. For the environment, the trade led to a net reduction in phosphorus loading into Lake Dillon. Although the Lake Dillon program was essentially unused for over 17 years, when the opportunity finally arrived, it made possible a wide range of environmental and economic benefits throughout the region.

NutrientNet

The principles of water quality trading are operationalized in the web site that is a companion to this guide, edu.NutrientNet.org. In NutrientNet participants play the role of either a wastewater treatment plant, a point source, or a farmer, a nonpoint source. The individuals are all placed in a watershed facing a required reduction in phosphorus loads. As in many real TMDLs, point sources are required to reduce their phosphorus load, while nonpoint sources are not required to reduce their loading. However, point source reductions are often more expensive than nonpoint source reductions. Hence, instead of implementing treatment practices at their wastewater treatment plants, point sources often find it cheaper to pay the nonpoint sources to reduce their loads. The credits that a point source purchases can be used to offset all or part of its load reduction requirement.

The educational version of NutrientNet is specifically designed for helping students learn about water quality trading. The original version, http://www.nutrientnet.org, was developed as a tool for use in real watershed with real water quality trading program. It has been developed for several real watersheds and offers an online environment in which trading can actually take place.

For more information about how to use NutrientNet, you should consult the various documents that can be downloaded at http://edu.nutrientnet.org/.

Endnotes

[1] The National Pollution Discharge Elimination System (NPDES) program of the USEPA sets specific pollutant discharge limits for all point sources discharging into U.S. waters. The program was developed in 1974 and has been expanded to include dischargers such as large concentrated animal feeding operations (CAFOs), municipal wastewater treatment facilities, and commercial and industrial facilities.

[2] This section is based on Woodward, 2003.

CHAPTER 4 MODELING NON-POINT SOURCE IMPACTS ON WATER QUALITY

As noted in the previous chapter, water quality policy depends upon the prediction of loads to the watershed and water quality trading requires the prediction of how loads change as a result of practice changes. Much progress has been made in the past 20 years in modeling non-point source impacts on water quality, largely due to technological advances in Geographic Information System (GIS) software and to the availability of large quantities of digital and remotely sensed data. Models of non-point source pollution vary in complexity and resolution, from field to watershed scale. For the purposes of water quality management, watershed-scale models are generally preferred because they provide the level of aggregation necessary to evaluate systemic impacts of land use on water quality. These models estimate pollutant loads to stream reaches or other water bodies at some time step (daily, monthly, annual). More recently developed models then apply another set of algorithms to predict the impact of those loads on water quality.

An excellent, though dated, review of non-point source models may be found in the USEPA guidance document “Compendium of Tools for Watershed Assessment and TMDL Development (USEPA, 1997). There are many computer models available for watershed modeling, but the three most common approaches are those based on the simple Universal Soil Loss Equation and its associated models, the more complex SEDMOD, and the very complex SWAT. Summaries of these three watershed modeling approaches are presented below.

Models Based on the Universal Soil Loss Equation (USLE)

The Universal Soil Loss Equation (Wiechmier and Smith, 1978) is an empirically based model for estimating annual sediment loads to the edge of fields. The USLE has been modified and applied in many other models, including the TMDL USLE (USEPA, 2001), the Agricultural Non-point Source Model (AgNPS), the Modified USLE (MUSLE), and the Revised USLE (RUSLE). The most broadly used of these models is RUSLE (USDA-ARS-NSL, 2003; Foster et al., 2003). There are two versions of RUSLE in use: RUSLE1, like the USLE, calculates annual sediment load to the edge of a farm field.; RUSLE2 uses a mathematical integration to estimate daily erosion values, which are then summed for an annual value. RUSLE requires relatively little data to run at the field or subwatershed level (Foster et al., 2003). They are easy to use, and are the most broadly applied of field sediment load estimators. A RUSLE based model is used to predict the nutrient loads from agricultural lands in NutrientNet.

The Spatially Explicit Delivery Model (SEDMOD).

SEDMOD is used to calculate spatially explicit sediment delivery ratios within a watershed, and is thus a useful tool for watershed management (Fraser, 1999). SEDMOD, is a GIS-based modeling approach that can provide more spatially explicit information on how much sediment is delivered to waterways from given points within a watershed. There are greater data requirements for this modeling approach, requiring information on soil texture, roughness, and slope, among other parameters.

The Soil and Water Assessment Tool (SWAT).

The SWAT model is a physically-based distributed-parameter river basin or watershed scale hydrological model developed by the USDA Agricultural Research Service (ARS) at Temple, TX (Arnold et al. 1994, Arnold et al. 1998). It predicts the impact of land management practices on water, sediment, and agricultural chemical yields in large, complex watersheds with varying soils, land use, and management conditions over long periods of time. It accounts for weather, surface runoff, return flow, percolation, evapotranspiration, transmission losses, pond and reservoir storage, crop growth and irrigation, groundwater flow, reach routing, nutrient and pesticide loading, and water transfer. The SWAT model is incorporated as a modeling tool in USEPA's Better Assessment Science Integrating Point and Non-point Sources (BASINS) program for use in the development of TMDLs as described in section 303(d) of the Clean Water Act. GIS interfaces have been developed for the SWAT model to facilitate the aggregation of input data for simulating watersheds. The ArcView interface developed for the SWAT model is used to prepare input data. This interface requires a land cover map, soils map, and the Digital Elevation Model (DEM) as spatial inputs (http://brc.tamus.edu/swat/). However, these interfaces are still too complex for stakeholder-driven scenario analysis.

Uncertainty and Complexity

In most watershed-level assessment and management activities the only thing we are sure of is that we are "in doubt" (Matlock et el., 1994; Hession et al., 1996a, 1996b, 1996c). There are many uncertainties inherent in such activities, including: monitoring/measurement error, model error, model input parameter errors, spatial variability, errors in spatial data layers within a GIS, the effects of aggregation of spatial data when modeling watersheds, and temporal variability. Making decisions taking account of these uncertainties is called risk assessment.

Many types of uncertainties have been identified in the literature including the inherent variability in natural processes, model uncertainty, and parameter uncertainty (Haan, 1989). Although not a desirable aspect of the modeling process, uncertainty and variation are ubiquitous in such analyses and should not be ignored. In the past, incorporating quantitative uncertainty analysis into modeling activities required special expertise and computing power. However, the accessibility of powerful personal computers and spreadsheet-based Monte Carlo analysis software make it possible for most assessors and managers to "honestly" incorporate uncertainty analysis into their analyses, thereby allowing for more knowledgeable decision making.

Beck (1987), in reviewing the analysis of uncertainty in water quality modeling, concluded that many of the larger, more complex water quality models can easily generate predictions with little or no confidence. Large mechanistic models are too complex to be subjected to adequate uncertainty analysis. Therefore, Reckhow (1994) suggested the use of simpler models with thorough uncertainty analysis. This phenomenon is referred to as the *Information Paradox* (Rowe, 1977): the more complex one's model becomes, the greater one's uncertainty will be because of the greater number of parameters to be estimated and the greater number of stochastic processes and model functions that must be included. Hence, as in NutrientNet, it is often preferred to use a relatively simple model over a highly complex ones.

Conclusions

Water quality policy is becoming increasingly complex and requires a wide range of tools including both economics and state-of-the-art bio-physical modeling. Yet despite all the progress that has been made, significant challenges remain. In particular, the control of pollution from non-point sources continues to be difficult because of the difficulty in predicting and monitoring their loads. Water quality models will help, but for the foreseeable future there can be no escaping the fact that water quality is not completely predictable. Policies such as water quality trading, therefore, must be designed taking this uncertainty into account.

REFERENCES

Adler, R. W., Jessica C. L., & Cameron, D. (1993). *The Clean Water Act 20 Years Later*. National Resources Defense Council. Washington, D.C.: Island Press.

Arbuckle, J. G. et al. 1993. *Environmental Law Handbook* (12^{th} ed.). Rockville, MD: Government Institute, Inc.

Arnold, J. G., Allen, P. N., & Bernhardt, G. (1993). A comprehensive surface-groundwater flow Model. *J. Hydrology* 142-47.

Arnold, J. G., Srinivasin, R., Muttiah, R. S., & Williams, J. R. (1998). Large area hydrologic modeling and assessment: Part I, Model development. JAWRA *34*(1), 73-89.

Beck, M. B. (1987). Water quality modeling: A review of the analysis of uncertainty. *Water Resources Research*, *23*(8), 1393-1442.

Faeth, P. (2000). *Fertile Ground: Nutrient Trading's Potential to Cost-Effectively Improve Water Quality*. Washington, D.C: World Resource Institute.

Foster, G. R., Toy, T., Renard, K. G. (2003). Comparison of the USLE, RUSLE1.06C, and Rusle2 for application to highly disturbed lands. *Proceedings of the First Interagency Conference On Research*. In *The Watersheds*, Oct. 27-30, 2003, Benson, AZ., 154-160.

Fraser, R. H. (1999). SEDMOD: A GIS-based delivery model for diffuse source pollutants (Doctoral dissertation, Yale University, Connecticut), 76-99.

Haan, C. T. (1989). Parametric uncertainty in hydrologic modeling. *Transactions of the ASAE* 32(1):137-146.

Hession, W.C., D.E. Storm, and C.T. Haan. (1996c). Two-phase uncertainty analysis: an example using the Universal Soil Loss Equation. *Transactions of the ASAE* 39(4):1309-1319.

Hession, W. C., Storm D. E., Haan C. T., Reckhow K.H., Smolen M.D., & Burks S. L. (1996b). Risk analysis of TMDLs in an uncertain environment. *Lake and Reservoir Management*, *12*(3), 331-347.

Hession, W. C., Storm D. E., Haan C. T., Burks S. L., & Matlock M. D. (1996a). A watershed-level ecological risk assessment methodology. *Water Resources Bulletin*, *32*(5), 1039-1054.

King, D and P. Kuch. Will Nutrient Credit Trading Ever Work?. *Environmental Law Institute*. Washington, DC. May, 2003.

Matlock M. D., Storm D. E., Sabbagh G. J., Burks S. L., Smolen M. D. & Haan C. T. (1994). An ecological risk assessment paradigm using the Spatially Integrated Model for Phosphorus Loading and Erosion (SIMPLE). *Journal of Aquatic Ecosystem Health*. 3:1-8.

Reckhow, K.H. (1994). Water quality simulation modeling and uncertainty analysis for risk assessment and decision making. *Ecological Modeling*, *72*, 1-20.

Rowe, W. D. (1977). *The Anatomy of Risk*. New York: John Wiley and Sons.

Stavins, R. (1998). What can we learn from the Grand Policy Experiment? Lessons from SO2 Allowance Trading. *Journal of Economic Perspectives*, *12*(3), 69-88.

Stevens Institute of Technology (2004). An investigation of water quality. *The global water sampling project*. Retrieved Sep 14, 2006, from: http://www.k12science.org/curriculum/waterproj/index.shtml

Tietenberg, T. (2003). *Environmental and Natural Resource Economics* (6^{th} ed.). Boston, MA: Addison Wesley.

U.S. Environmental Protection Agency (USEPA), (1997). *Compendium of tools for watershed assessment and TMDL development, EPA841-B-97-006*. Washington, DC: USEPA Office of Water

U.S. Environmental Protection Agency (USEPA), (2000). *Monitoring and assessing water quality*.

Retrieved Feb 25, 2006, from http://www.epa.gov/305b/2000report/

U.S. Environmental Protection Agency (USEPA), (2001). *TMDL USLE*. Retrieved Sep 14, 2006, from http://www.epa.gov/ceampubl/swater/usle/

U.S. Environmental Protection Agency (USEPA), (2004a). *Water quality criteria*. Retrieved Sep 14, 2006, from http://epa.gov/waterscience/criteria/dissolved/dofacts.html

U.S. Environmental Protection Agency (USEPA), (2004c). *Clean air markets - allowance trading*. Retrieved Sep 14, 2006 from http://www.epa.gov/AIRMARKET/trading/index.html

U.S. Environmental Protection Agency (USEPA). (2004d). *Final water quality trading policy*. Office of Water. Retrieved Oct 27, 2004, from: http://www.epa.gov/owow/watershed/trading/finalpolicy2003.html

Wishmeier, W., & Smith. D. (1978). Predicting rainfall erosion losses. USDA *Agriculture Handbook* #537. Washington, DC: USDA,.

Woodward, R.T, Kaiser, R.A. and A.M.B. Wicks. (2002). The structure and practice of water quality trading markets. *Journal of the American Water Resources Association*, *38*(4), 967-979.

Woodward, R. T. (2003). Lessons about effluent trading from a single trade. *Review of Agricultural Economics*, *25*: 235-245.