**Economics of Water Quality Protection From Nonpoint Sources: Theory and Practice.** By Marc O. Ribaudo, Richard D. Horan, and Mark E. Smith. Resource Economics Division, Economic Research Service, U.S. Department of Agriculture. Agricultural Economic Report No. 782.


## Abstract

Water quality is a major environmental issue. Pollution from nonpoint sources is the single largest remaining source of water quality impairments in the United States. Agriculture is a major source of several nonpoint-source pollutants, including nutrients, sediment, pesticides, and salts. Agricultural nonpoint pollution reduction policies can be designed to induce producers to change their production practices in ways that improve the environmental and related economic consequences of production. The information necessary to design economically efficient pollution control policies is almost always lacking. Instead, policies can be designed to achieve specific environmental or other similarly-related goals at least cost, given transaction costs and any other political, legal, or informational constraints that may exist. This report outlines the economic characteristics of five instruments that can be used to reduce agricultural nonpoint source pollution (economic incentives, standards, education, liability, and research) and discusses empirical research related to the use of these instruments.



**Keywords:** water quality, nonpoint-source pollution, economic incentives, standards, education, liability, research.



Washington, DC 20036-5831 November 1999

# Contents

# Executive Summary

## What Is the Problem?

The quality of the Nation's surface water has improved since 1972's Clean Water Act, primarily through reductions in pollution from industrial and municipal sources. However, water quality problems remain, especially those associated with nonindustrial sources. The latest EPA Water Quality Inventory reports that, of the water resources assessed by the States, more than one-third of the river miles, lake acres, and estuary square miles suffer some degree of impairment.

Water pollution may be categorized into two types. Point-source pollution enters water resources directly through a pipe, ditch, or other conveyance. Industrial and municipal discharges fall into this category. Nonpoint-source pollution enters water diffusely in the runoff or leachate from rain or melting snow and is often a function of land use. Nonpoint-source pollution has been identified as a major reason for remaining U.S. water quality problems. Despite some progress in reducing agricultural production practices believed harmful to water quality, agriculture is generally recognized as the largest contributor to nonpoint-source water pollution in the United States.

Primary agricultural pollutants are sediment, nutrients, pesticides, salts, and pathogens. A U.S. Geological Survey (USGS) study of agricultural land in watersheds with poor water quality estimated that 71 percent of U.S. cropland (nearly 300 million acres) is located in watersheds where the concentration of at least one of four common surface-water contaminants (nitrate, phosphorus, fecal coliform bacteria, and suspended sediment) exceeded criteria for supporting water-based recreation activities. Well-water sampling by EPA and USGS has found evidence of agricultural pesticides and nitrogen in groundwater resources, possibly threatening water supplies in some areas. Comprehensive estimates of the damages from agricultural pollution are lacking, but soil erosion alone is estimated to cost water users $2 billion to $8 billion annually.

## Why Are Nonpoint Pollution Control Policies Needed and What Are the Issues Involved?

Nonpoint-source water pollution is an externality to the production process. Externalities exist when some of the consequences of production (pollution's imposing costs on others) are not considered when production decisions are made. The result is a misallocation of resources from society's perspective.

A fundamental goal of environmental policy is to induce polluters to explicitly consider the costs they impose on society through their production-related activities. An ideal goal of policy is to maximize the expected net economic benefits to society from pollution control, also known as the economically efficient or first-best outcome. Designing policies to achieve efficiency, however, is often impossible because the relationship between economic damages and nonpoint pollution is seldom known. Instead, policies can be designed to achieve specific environmental goals (such as reducing ambient pollution levels or reducing fertilizer applications in a region) at least cost, given the policy instruments available to a resource management agency, relevant policy transactions costs, and any other political, legal, or informational constraints that may exist. Such outcomes are often referred to as cost-effective or second-best.

The process of designing comprehensive policies for controlling nonpoint pollution therefore consists of defining appropriate policy goals, choosing appropriate instruments, and setting these instruments at levels that will achieve the goals at least cost. There are difficulties associated with each of these aspects due to the complex physical nature of the nonpoint pollution process.

Nonpoint emissions (runoff) cannot be measured at reasonable cost with current monitoring technologies because they are diffuse (i.e., they move off the fields in a great number of places) and are impacted by random events such as weather. In addition, the process by which runoff is transported to a water body where it creates economic damages is also impacted by random events. The random nature of these physical processes creates some significant limitations in the way that policy goals with good economic properties are defined, and in the types of policy tools that can be used to attain a cost-effective outcome.

Finally, runoff depends on many site-specific factors. The better that policies and goals can address these site-specific factors, the more efficient nonpoint policies will be. However, obtaining the appropriate information to adequately design and implement policies that address site-specific factors may be quite costly. These costs may limit the types of policies (e.g., to those that are more uniformly applied and informationally less intensive) that can be used to control nonpoint pollution.

## What Types of Policy Instruments Can Be Applied to Nonpoint-Source Pollution?

Five classes of policy instrument have either been applied to nonpoint-source pollution, or are feasible tools. These are economic incentives, standards, education, liability, and research. In evaluating a tool's potential, a number of important economic, distributional, and political characteristics are considered. These include economic performance (ability to achieve a goal at least cost), administration and enforcement costs, flexibility (able to provide effective control in the face of changing economic and environmental conditions), incentives for innovation, and political feasibility.

*Economic incentive-based* instruments include performance incentives (taxes on runoff or ambient water quality), design incentives (taxes or subsidies on inputs and technology), and market-based approaches such as point/nonpoint trading (allowing different sources to trade abatement allowances). Ideally, incentives are directed at an aspect of the pollution process (the instrument base) that is closest to the water quality problem, such as ambient water quality or runoff into a stream (e.g., a runoff tax or subsidy). However, because nonpoint-source discharges cannot be observed, runoff-based instruments are currently infeasible. In this report, we show that the most practical incentive-based instruments are design incentives (including expected runoff incentives that use runoff models), and market-based approaches (also based on design elements). Incentive policies have generally not been applied to agricultural nonpoint-source pollution. Cost-shares and other financial incentives offered by USDA are not subsidies in the traditional sense, in that they are only offered over the short term.

*Standards* use the regulatory system to mandate that producers meet a particular environmental goal, or that they adopt more socially efficient management practices. In theory, standards can be applied to performance measures, such as runoff or ambient quality, or to inputs and technology. As with incentives, performance-based standards are generally infeasible. Design-based standards, which are feasible, include standards based on expected runoff (which is estimated with information on input use and technology choice through the use of simulation models) and standards based more directly on input use and technology choices.

Design-based standards are being widely applied to agricultural nonpoint-source pollution problems. Some examples include the required use of best management practices on cropland, mandatory establishment of riparian buffer strips, and restrictions on where and at what rates agricultural chemicals can be applied.

*Liability* rules can be used to guide compensation decisions when polluters are sued for damages in a court of law. Such rules, although they are employed only after damages occur and if victims are successful in their suit, can theoretically provide ex ante incentives for polluters to account for the environmental consequences of their actions. Liability rules can be developed under two different frameworks: strict liability and negligence. Polluters are held absolutely liable for payment of any damages that occur under strict liability. Alternatively, polluters are liable under a negligence rule only if they failed to act with the "due standard of care."

In theory, an efficient level of pollution control can be achieved for each type of rule. In practice, however, the characteristics of nonpoint-source pollution limit the feasibility of liability tools for achieving efficient control. Liability depends on being able to identify the individual sources of pollution when damages occur. The inability to trace nonpoint pollution back to its source greatly weakens the effectiveness of liability. In addition, liability rules that are based on performance measures require polluters to understand how their choices impact the performance measures. If these impacts are difficult to predict or require an extensive amount of information, liability rules will be less than effective in promoting more efficient production. Liability rules are probably best suited for the control of pollution related to the use of hazardous materials, or for nonfrequent occurrences such as accidental chemical spills. Liability is currently being used in some States to protect groundwater supplies from agricultural chemicals.

*Education* provides producers with information on how to farm more efficiently with current technologies (minimizing excess use of chemicals, for example), or about new technologies that generate less pollution and are more profitable (conservation tillage). While such "win-win" solutions to water quality problems are attractive, education cannot be considered a strong tool for water quality protection. Its success depends on alternative practices being more profitable than conventional practices, or that producers value cleaner water enough to accept potentially lower profits. Evidence from USDA education programs suggests that net returns are the predominant concern of producers when adopting alternative management practices. Producers have not exhibited interest, in general, in adopting practices that do not benefit them personally. In other words, they do not voluntarily account for any externalities they create. A more appropriate role for education is as a support tool for other policies. Education can shorten the time it takes producers to successfully adopt alternative practices promoted through other policies. Education is widely used by USDA to promote the adoption of alternative management practices.

*Research and development* can be an important component of a policy for reducing agricultural nonpoint-source pollution because it provides producers and society with more efficient ways of meeting environmental goals. However, producers and private firms will necessarily underinvest in research and development for water quality-improving innovations. Not all the benefits from research result in economic returns to investors.

Public sector involvement is necessary either to carry out this research or to provide producers and the private sector with incentives that result in more efficient research investments. Finally, research cannot independently provide a solution to water quality problems. Research cannot make producers account for the externalities resulting from their production practices. Instead, it serves as a valuable component of other approaches by expanding the set of alternative production practices.

## What Is the Guidance for Nonpoint-Source Policy?

The characteristics of nonpoint-source pollution currently render performance-based policies infeasible. Education and research can be valuable in a support role, but cannot stand alone. This leaves design-based policies such as design standards and design incentives (including market-based approaches) as the most viable options. The characteristics of nonpoint-source pollution and the diversity of resource conditions important to agriculture rule against a single tool being applied to all problems. For example, a nitrates-in-groundwater problem might require a combination of fertilizer bans in well recharge areas, reduced application rates elsewhere, the use of cover crops to soak up nitrogen remaining in the soil after harvest, and the use of long-term easements to retire marginal cropland. The tool or combination of tools best suited for a particular problem is an empirical issue based on policy goals, local conditions, and the costs of acquiring information. Policies designed to control the quality of expected or predicted runoff have some of the desirable characteristics of performance-based policies, but depend on models for estimating runoff. Development of models that can estimate agricultural pollutant flows in a variety of geographic and agronomic settings would greatly improve effectiveness of nonpoint-source control policies.

# Economics of Water Quality Protection From Nonpoint Sources

## Theory and Practice

**Marc O. Ribaudo**
**Richard D. Horan**
**Mark E. Smith**

## Introduction

The harmful consequences of farm production on water quality include soil erosion; runoff into rivers and streams of fertilizers, animal waste, and pesticides; and leaching into groundwater of nutrients and pesticides. However, agricultural pollution is only one source of water quality problems; others include discharges from industry and municipal sewage treatment plants, urban runoff, and atmospheric deposition (delivery by wind and rain). Still, agriculture is identified as a major contributor to pollution of the Nation's surface waters (EPA, 1998a).

Public concern over the degradation of water resources has led to a number of Federal, State, and local policies and programs for protecting and improving water quality. The response has been multifaceted. Both regulatory and voluntary programs have been administered by a variety of Federal, State, and local agencies. On February 19, 1998, the White House released the Clean Water Action Plan. The plan states that:

> After 25 years of progress, the nation's clean water program is at a crossroads. Implementation of the existing programs will not stop serious new threats to public health, living resources, and the nation's waterways, particularly from polluted runoff. These programs lack the strength, resources, and framework to finish the job of restoring rivers, lakes, and coastal areas. To fulfill the original goal of the Clean Water Act—fishable and swimmable water for every American—the nation must chart a new course to address the pollution problems of the next generation. (EPA, USDA, 1998, pg. i).

Controlling water pollution can follow many courses. Economics has an important, if not vital, role to play in identifying policy strategies that can enhance water quality at least cost. An economic framework can coordinate policy formulation among different levels of government and help to unify policies across regions.

Reducing pollution requires changing the behavior of polluters. Since polluters are already operating within an economic framework (the profit-maximizing one), water quality protection policies can be seen as altering some of the economic variables a polluter considers when making everyday production decisions.

On the other hand, economics also determines the optimal level of water quality protection. Society does not benefit from overly stringent or costly water quality goals. Measuring the benefits of water quality protection to water users in economic terms is often difficult, since many benefits occur outside of easily observable market conditions. Even where water quality impacts on markets are observed, it can be difficult to ascertain just how water pollution affects the ability of a resource to provide economic goods. Nevertheless, information on benefits is essential to developing socially optimal water quality protection policies.

In this report, we review alternative policy tools for addressing nonpoint-source pollution. Much progress has been made in controlling pollution from point sources, such as factories and municipal sewage treatment plants. However, nonpoint-source pollution is much more complicated and elusive than point-source pollution, and the tools developed for controlling one do not necessarily apply to the other. We first present what is currently known about the quality of the Nation's water resources and agriculture's contributions to existing problems. The second chapter presents some guidelines for efficient policy design. We then review some issues surrounding policy development and implementation, including the characteristics of nonpoint-source pollution and the level of government—Federal or local—best suited to addressing those problems. The next five chapters cover five classes of policy tools: economic incentives, standards, liability, education, and research and development. Finally, we suggest the roles of different policy instruments in a national strategy to control nonpoint-source pollution, and identify additional research needed to improve such a strategy.

# Chapter 1

# Current Water Quality Conditions and Government Programs To Protect Water Quality

*The quality of the Nation's water is an important environmental issue. While water quality laws passed since 1972 have resulted in some improvements, many water quality problems remain. The latest EPA Water Quality Inventory reports that, of the water resources assessed by the States, more than one-third of the river miles, lake acres, and estuary square miles are impaired to some degree. Nonpoint-source pollution has been identified as a major reason for these problems, with agriculture a major contributor. Agricultural pollutants include sediment, nutrients, pesticides, salinity, and pathogens. Comprehensive estimates of the damages from agricultural pollution are lacking, but soil erosion alone is estimated to cost water users $2 billion to $8 billion annually. Federal and State programs rely heavily on economic and educational tools to deal with water quality problems. Inadequate water quality monitoring hinders use of a full range of policy instruments to deal with nonpoint-source water pollution.*

## Water Quality in the United States

The Nation's surface-water quality has improved since 1972's Clean Water Act, primarily through reductions in pollution from industrial and municipal sources. No longer are there news stories of the Cuyohoga River catching fire, or Lake Erie being biologically dead. Indeed, we read stories about increasing recreational use of major rivers such as the Potomac, Delaware, and Hudson, even close to major urban areas. However, water quality problems remain, especially those associated with nonindustrial sources. We now read of microbe-related fish kills in nutrient-enriched waters, the presence of pesticides in drinking water, and the degradation by nutrients of important national resources such as the Gulf of Mexico, Chesapeake Bay, and the Everglades. In 1998, the White House called for a shift in national water quality policy to address more effectively the problems caused by nonpoint-source pollution (EPA, USDA, 1998).

Water pollution may be categorized by two types of sources. Point sources discharge effluent directly into water resources through an identifiable pipe, ditch, or other conveyance. Industrial and municipal discharges fall into this category. Nonpoint-source pollution (NPS) enters water diffusely in the runoff or leachate from rain or melting snow, and is often a function of land use. Agriculture is generally recognized as the largest contributor to NPS water pollution in the United States (EPA, 1998a). Animal waste and certain farm practices (soil tillage, use of chemicals, use of irrigation) are the major sources of pollutants such as sediment, nutrients, pesticides, salts, and pathogens.

The first part of this chapter presents what is known about the current condition of the Nation's water resources. The second section summarizes agriculture's contribution to specific water quality problems. The costs of water pollution are then presented, along with Federal and State programs to address water pollution. The chapter concludes with a discussion of how deficiencies in current water quality data affect water quality policies.

### Surface Water

Since the passage of the Clean Water Act (33 U.S.C. §§ 1288, 1329) in 1972, water quality has improved largely through reductions in toxic and organic chemical loadings from point sources. Discharges of toxic pollutants have been reduced by an estimated billion pounds per year (Adler, 1994). Rivers affected by sewage treatment plants show a consistent reduction in

**Table 1-1—Status of the Nation's surface-water quality, 1990-96**

| | Rivers | | | | Lakes[1] | | | | Estuaries | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Item | 1990 | 1992 | 1994 | 1996 | 1990 | 1992 | 1994 | 1996 | 1990 | 1992 | 1994 | 1996 |
| | *Percent of total water** | | | | | | | | | | | |
| Water systems assessed | 36 | 18 | 17 | 19 | 47 | 46 | 42 | 40 | 75 | 74 | 78 | 72 |
| | *Percent of assessed waters* | | | | | | | | | | | |
| Meeting designated uses[2]: | | | | | | | | | | | | |
| Supporting | 69 | 62 | 64 | 64 | 60 | 56 | 63 | 61 | 67 | 68 | 63 | 62 |
| Partially supporting[3] | 21 | 25 | 22 | 36 | 19 | 35 | 28 | 39 | 25 | 23 | 27 | 38 |
| Not supporting | 10 | 13 | 14 | | 21 | 9 | 9 | | 8 | 9 | 9 | |
| Clean Water Act goals: Fishable | | | | | | | | | | | | |
| Meeting | 80 | 66 | 69 | 68 | 70 | 69 | 69 | 69 | 77 | 78 | 70 | 69 |
| Not meeting | 19 | 34 | 31 | 31 | 30 | 31 | 31 | 31 | 23 | 22 | 30 | 30 |
| Not attainable | 1 | - | - | - | 0 | - | - | - | - | 0 | 0 | 0 |
| Clean Water Act goals: Swimmable | | | | | | | | | | | | |
| Meeting | 75 | 71 | 77 | 79 | 82 | 77 | 81 | 75 | 88 | 83 | 85 | 84 |
| Not meeting | 15 | 20 | 23 | 20 | 18 | 22 | 19 | 25 | 12 | 17 | 15 | 16 |
| Not attainable | 10 | 9 | - | - | - | - | - | - | - | 0 | - | - |

- = less than 1 percent of assessed waters.
[1] Excluding Great Lakes.
[2] Supporting - water quality meets designated use criteria; partially supporting - water quality fails to meet designated use criteria at times; not supporting - water quality frequently fails to meet designated use criteria.
[3] In 1996, the categories "Partially supporting" and "Not supporting" were combined.
* Miles of rivers, acres of lakes, square miles of estuaries.

Source: Environmental Protection Agency, National Water Quality Inventories (1992b, 1994b, 1995, 1998a).

ammonia between 1970 and 1992 (Mueller and Helsel, 1996). The percentage of the U.S. population served by wastewater treatment plants increased from 42 percent in 1970 to 74 percent in 1985 (Adler, 1994). A widely scattered surface-water monitoring network has shown national reductions in fecal bacterial and phosphorus concentrations (Knopman and Smith, 1993; Smith, Alexander, and Lanfear, 1993; Lettenmaier, Hooper, Wagoner, and Faris, 1991; Mueller and Helsel, 1996). Case studies, opinion surveys, and anecdotal information suggest that these reductions in pollutants have improved the health of aquatic ecosystems in many basins, particularly near urban areas (Knopman and Smith, 1993). However, challenges to water quality remain, including continuing discharges of pollutants from a growing population and economy, inadequate discharge permit requirements in some States, violations of permits issued, and pollution from nonpoint sources.

The most recent EPA Water Quality Inventory reports indicate the nature of water quality impairments (table 1-1) (EPA, 1998a). The Water Quality Inventory is prepared with information contained in biennial reports from the States, required by the Clean Water Act, on the status of their surface-water resources (known as Section 305(b) reports). In 1996, 36 percent of river miles, 39 percent of lake acres (excluding the Great Lakes), and 38 percent of estuary square miles were found to not fully support the uses for which they were designated by States under the Clean Water Act (see box 1.1). States reported that agriculture is the leading source of impairment in the Nation's rivers and lakes, and a major source of impairment in estuaries.

While many agencies and organizations assess water quality, only the 305(b) reports provide a snapshot of how well waters across the Nation meet designated uses (see box 1.2). However, 305(b) data are not gath-

**Box 1.1—How Is Water Quality Defined?**

The Clean Water Act (passed in 1972 as the Federal Water Pollution Control Act) defines water quality in terms of designated beneficial uses with numeric and narrative criteria that support each use. Designated beneficial uses are the desirable uses that water quality supports. Examples are drinking water supply, primary-contact recreations, and aquatic life support. Numeric water quality criteria establish the minimum physical, chemical, and biological parameters required for water to support a beneficial use. Physical and chemical criteria may set maximum concentrations of pollutants, acceptable ranges of physical parameters, and minimum concentrations of desirable parameters, such as dissolved oxygen. Biological criteria describe the expected attainable community attributes and establish values based on measures such as species richness, presence or absence of indicator species, and distribution of classes of organisms (EPA, 1994b). Narrative water quality criteria define conditions and attainable goals that must be maintained to support a designated use. Narrative biological criteria describe aquatic community characteristics expected to occur within a water body.

The Clean Water Act allows jurisdictions to set their own standards but requires that all beneficial uses and their criteria comply with the goals of the Act. At a minimum, beneficial uses must provide for the "protection and propagation of fish, shellfish, and wildlife" and provide for "recreation in and on the water" (fishable and swimmable) (U.S. Congress, PL 92-500, 1972, p. 31). The Act prohibits waste assimilation as a beneficial use.

Source: U.S. Congress, PL 92-500, 1972.

ered in a consistent manner from one State to another, and often are not based on actual monitoring. Only a portion of water bodies are actually monitored in any given year (ranging from 19 percent of rivers and streams to 94 percent of Great Lakes shoreline in 1996), so variations in estimates between years could be due to changes in actual water quality, changes in the water bodies sampled, or changes in assessment protocols. These data cannot therefore be used to identify trends.

Nationwide, about one-third of surface waters are deemed impaired, but large, regional problems exist. These include:

- The Great Lakes show only 3 percent of the assessed shoreline miles (with 94 percent assessed) fully supporting designated uses (EPA, 1998a). Fish consumption is the designated use most frequently impaired. Most of the shoreline is polluted with toxic chemicals, primarily polychlorinated biphenyls (PCB's), mercury, pesticides, and dioxins that are often found in fish samples. Atmospheric deposition of toxics (delivery by wind or rain), point sources, and contaminated sediment are the leading sources of impairment.

- The Chesapeake Bay, the largest estuary in the world, has seen water quality degrade due primarily to elevated levels of nitrogen and phosphorus (EPA, 1998a). An aggressive pollution control program has reduced phosphorus, but nitrogen concentrations have largely remained unchanged, leaving the bay overenriched. Excess nitrogen and phosphorus promote algae growth that clouds the water and reduces oxygen levels. Excessive nutrient levels in tributaries of the Bay are believed responsible for the outbreak of the micro-organism *Pfiesteria*, which led to large fish kills in 1997 (Mlot, 1997). Shellfish harvests have declined dramatically in recent years, and poor water quality is believed to be an important contributing factor (State of Maryland, 1984).

- The Gulf of Mexico has seen since 1993 a doubling in the size of an oxygen-deficient "dead" zone to 7,000 square miles (Rabalais, Turner, and Wiseman, 1997). The primary cause is believed to be increased levels of nitrates carried to the gulf by the Mississippi and Atchafalaya Rivers. The amount of nitrate discharged into the gulf has increased threefold since 1954 (Goolsby and Battaglin, 1997). A major source of nitrates is fertilizers from the Upper Mississippi Basin (Antweiler, Goolsby, and Taylor, 1995).

## Ground Water

Groundwater quality in the United States is not well known. Unlike surface water, no comprehensive groundwater monitoring system exists. However, many States report on the general quality of their groundwater resources in their section 305(b) reports.

## Box 1.2—Assessing Water Quality

Many Federal, State, and local agencies and private groups monitor water quality (EPA, 1997c). The U.S. Geological Survey (USGS) monitors surface and ground water extensively. For example, under its National Stream Quality Accounting Network, 618 watersheds of major U.S. rivers and streams are monitored for physical characteristics (e.g., stream flow, temperature) and quality characteristics (e.g., nutrient levels). The USGS National Water Quality Assessment Program uses a regional focus to study status and trends in water, sediment, and biota in selected major watersheds. The Environmental Protection Agency (EPA) provides grants for water quality monitoring, or, in some cases, conducts monitoring itself. Under its National Monitoring Program, EPA attempts to obtain long-term data on the effectiveness of nonpoint-source pollution control measures. The Environmental Monitoring and Assessment Program is designed to provide information on status and trends of selected waters for a variety of ecosystems. Other Federal agencies involved in water quality monitoring include the U.S. Fish and Wildlife Service, the National Oceanic and Atmospheric Administration, and the U.S. Army Corps of Engineers. In some cases, other agencies and groups may receive Federal support for monitoring, or they may conduct such activities for their own uses.

However, using monitoring data to assess water quality at a national level is not a simple exercise. Water quality varies by time, location, and depth (e.g., shallow or deep portion of an aquifer or reservoir). Further, water quality is composed of a variety of characteristics, the importance of which will vary with the desired use of the water (e.g., dissolved oxygen concentration to support aquatic life; nitrate or pesticide concentrations that may violate drinking water standards; the presence of pathogens that would inhibit recreational uses). In many cases, monitoring is often done to study only one or a few components of water quality, or a specific problem, and might not address other quality questions. USGS reports status and trends of specific characteristics of water in which one may be interested, but does not weight the characteristics to develop an aggregate measure. EPA, in its biennial report to Congress on the Nation's water quality, draws from the States' assessments of how well waters meet their designated uses to report an aggregate measure of water quality in different water sources (e.g., rivers, lakes, estuaries), though there is no standardization across the States.

Of 38 States that reported overall groundwater quality in 1992, 29 judged their groundwater quality to be good or excellent (EPA, 1994b). Generally, States report that contamination of ground water is localized. In 1994, over 45 States reported that pesticide and fertilizer applications were sources of groundwater contamination (EPA, 1995). Other indications of groundwater quality come from the EPA's National Survey of Pesticides in Drinking Water Wells, conducted in 1988-90. The survey provided the first national estimates of the frequency and concentration of nitrates and pesticides in community water system wells and rural domestic drinking water wells.

## Agricultural Pollutants

Both natural and human-caused sources of pollutants affect the Nation's water resources. Anthropogenic sources include point sources, such as industrial and municipal discharges, and nonpoint sources such as agriculture, forestry, construction, and urban runoff.

Agricultural pollutants include sediment, nutrients (nitrogen and phosphorus), pesticides, salts, and pathogens. While farmers do not intend for these materials to move from the field or enterprise, they often do. For example, as much as 15 percent of the nitrogen fertilizer and up to 3 percent of pesticides applied to cropland in the Mississippi River Basin make their way to the Gulf of Mexico (Goolsby and Battaglin, 1993). A U.S. Geological Survey (USGS) study of agricultural land in watersheds with poor water quality estimated that 71 percent of U.S. cropland (nearly 300 million acres) is located in watersheds where the concentration of at least one of four common surface-water contaminants (dissolved nitrate, total phosphorus, fecal coliform bacteria, and suspended sediment) exceeds criteria for supporting water-based recreation (Smith, Schwarz, and Alexander, 1994).

### Sediment

Disturbing the soil through tillage and cultivation and leaving it without vegetative cover increases the rate of soil erosion. Dislocated soil particles can be carried in runoff water and eventually reach surface-water resources, including streams, rivers, lakes, reservoirs, and wetlands.

Sediment causes various damage to water resources and to water users. Accelerated reservoir siltation reduces the useful life of reservoirs. Sediment can clog roadside ditches and irrigation canals, block navigation channels, and increase dredging costs. By raising stream beds and burying streamside wetlands, sediment increases the probability and severity of floods. Suspended sediment can increase the cost of water treatment for municipal and industrial water uses. Sediment can also destroy or degrade aquatic wildlife habitat, reducing diversity and damaging commercial and recreational fisheries. Siltation is the leading pollution problem in U.S. rivers and streams (EPA, 1998a). Sediment damages from agricultural erosion have been estimated to be between $2 billion and $8 billion per year (Ribaudo, 1989). These estimates include damages or costs to navigation, reservoirs, recreational fishing, water treatment, water conveyance systems, and industrial and municipal water use.

Trends in erosion losses and instream sediment concentration seem to show improvements in recent years. The National Resources Inventory reports that the average rate of sheet and rill erosion on cropland declined by about one-third between 1982 and 1992. In most regions, the USGS found that suspended sediment concentrations trended slightly downward over the 1980's, particularly in the Ohio-Tennessee, and Upper and Lower Mississippi regions (table 1-2) (Smith, Alexander, and Lanfear, 1993). Areas characterized by corn and soybean production and mixed crops had the greatest downward trends. Soil conservation efforts over the past 10 years, particularly the Conservation Reserve Program and Conservation Compliance, likely played a role (USDA, ERS, 1997). Table 1-3 shows estimated benefits of soil conservation programs to be on the order of several hundred million dollars to billions of dollars over the life of the conservation practices adopted.

**Table 1-2—Trends in concentrations of agricultural water pollutants in U.S. surface waters, 1980-90**

| Water resources region | Nitrate | Total phosphorus | Suspended sediment |
|---|---|---|---|
| | *Average percentage change per year* | | |
| North Atlantic | * | -1.4 | -0.4 |
| South Atlantic-Gulf | * | 0.1 | 0.2 |
| Great Lakes | * | -3.3 | 0.5 |
| Ohio-Tennessee | * | -1.0 | -1.3 |
| Upper Mississippi | -0.4 | -1.2 | -1.3 |
| Lower Mississippi | -1.6 | -3.8 | -1.2 |
| Souris-Red-Rainy | * | -0.8 | 1.2 |
| Missouri | * | -1.7 | -0.2 |
| Arkansas-White-Red | * | -3.1 | -0.7 |
| Texas-Gulf-Rio Grande | * | -0.9 | -0.6 |
| Colorado | * | -2.4 | -0.8 |
| Great Basin | * | -2.7 | -0.2 |
| Pacific Northwest | * | -1.7 | -0.1 |
| California | * | -1.4 | -0.6 |

* Between -0.1 and 0.1.
Source: Smith, Alexander, and Lanfear, 1993.

## Nutrients

Nutrients, chiefly nitrogen, potassium, and phosphorus, promote plant growth. About 11 million tons of nitrogen, 5 million tons of potash (the primary chemical form of potassium fertilizer), and 4 million tons of phosphate (the primary chemical form of phosphorus fertilizer) are applied each year to U.S. cropland (USDA, ERS, 1997). Nutrients can enter water resources three ways. *Runoff* transports pollutants over the soil surface by rainwater, melting snow, or irrigation water that does not soak into the soil. Nutrients move from fields to surface water while dissolved in runoff water or adsorbed to eroded soil particles. *Run-in* transports chemicals directly to ground water through sinkholes, porous or fractured bedrock, or poorly constructed wells. *Leaching* is the movement of pollutants through the soil by percolating rain, melting snow, or irrigation water.

Of the three nutrients, nitrogen and phosphorus can cause quality problems when they enter water systems. Nitrogen, primarily found in the soil as nitrate, is easily soluble and is transported in runoff, in tile drainage, and with leachate. Phosphate is only moderately soluble, and relative to nitrate, is not very mobile in soils. However, erosion can transport considerable amounts of sediment-adsorbed phosphate to surface waters. If soils have been overfertilized, rates of dissolved phosphorus losses in runoff will increase due to the buildup of phosphates in the soil.

Nitrogen and phosphorus from agriculture accelerate algal production in receiving surface water, resulting in a variety of problems, including clogged pipelines, fish kills, and reduced recreational opportunities (EPA, 1998a). Nitrogen is primarily a problem in brackish or salt water, where it is the limiting nutrient, while phosphorus is primarily a problem in freshwater. EPA reports that nutrient pollution is the leading cause of water quality impairment in lakes and estuaries, and is the second leading cause in rivers (EPA, 1998a). Increases in the occurrence of harmful algal blooms in coastal waters have been attributed to nutrients from human-caused sources, including fertilizers (Boesch and others, 1997).

Besides harming aquatic ecosystems, nitrate is also a potential human health threat. The EPA has established a maximum contaminant level (MCL, a legal maximum long-term exposure) in drinking water of 10 mg/liter. Nitrate can be converted to nitrite in the gastrointestinal tract. In infants, nitrite may cause methemoglobinemia, otherwise known as "blue-baby syndrome," which prevents the transport of sufficient oxygen in the bloodstream. Public water systems that violate the MCL must use additional treatment to bring the water they provide into compliance, though exemptions are specified (42 U.S.C. §300g).

Data (from USGS monitoring stations) on nutrients in surface waters over the 1980's show different trends for nitrate and phosphorus (table 1-2) (Smith, Alexander, and Lanfear, 1993). Nitrate, in general, showed no statistically significant trend, which differs from the rise noted during 1974-81 (Smith, Alexander, and Wolman, 1987). This follows the pattern of agricultural nitrogen use, which rose sharply during the 1970's, peaked in 1981, and then stabilized. Phosphorus in water during the 1980's continued a decline noted in the 1970's, likely due to improved wastewater treatment, decreased phosphorus content of detergents, reduced phosphorus fertilizer use, and reduced soil erosion. Indeed, the rate of phosphorus decline in water in cropland areas was more than twice that in urban areas (Smith, Alexander, and Lanfear, 1993).

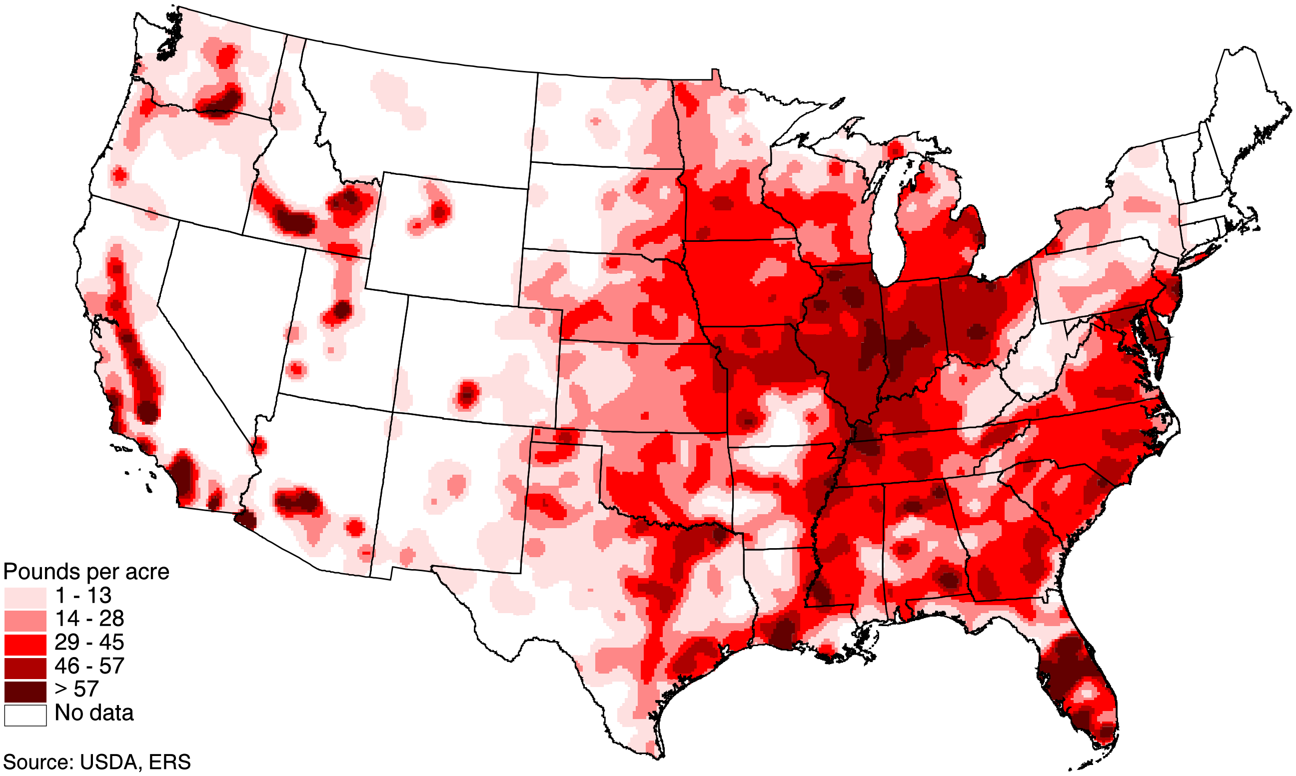
Exposure to nitrate in drinking water is chiefly a concern to those whose source water is ground water, which generally has higher nitrate concentrations than surface water (Mueller and others, 1995). From its 1988-90 national survey of drinking water wells, the EPA found nitrate in more than half of the 94,600 community water system wells (CWS) and almost 60 percent of the 10.5 million rural domestic drinking water wells, making nitrate the most frequently detected chemical in well water (EPA, 1992a). However, only 1.2 percent of the CWS's and 2.4 percent of the rural domestic wells were estimated to contain levels above the MCL. About 3 million people (including 43,500 infants) using water from CWS's and about 1.5 million people (including 22,500 infants) using rural wells are exposed to nitrate at levels above the MCL (EPA, 1992a).

A 1991 USGS study of nitrate in near-surface aquifers in the midcontinental United States detected nitrate in 59 percent of the samples taken (Kolpin, Burkart, and Thurman, 1994). Concentrations greater than the MCL were found in 6 percent of the samples. Statistical analyses indicated that the frequency of samples having concentrations greater than 3 mg/l (believed to be the maximum level from natural sources) was positively related to the proximity of agricultural land, to the use of irrigation, and to fertilizer application rates.

More recently, in a study of well water samples in 18 USGS National Water Quality Assessment Program study units, USGS found that the MCL was exceeded in about 1 percent of CWS's and 9 percent of rural domestic wells (Mueller and others, 1995). About 16 percent of domestic wells under agricultural land exceeded the MCL in selected watersheds, with particularly high proportions exceeding the MCL in the Northern Plains (35 percent) and the Pacific (27 percent) regions.

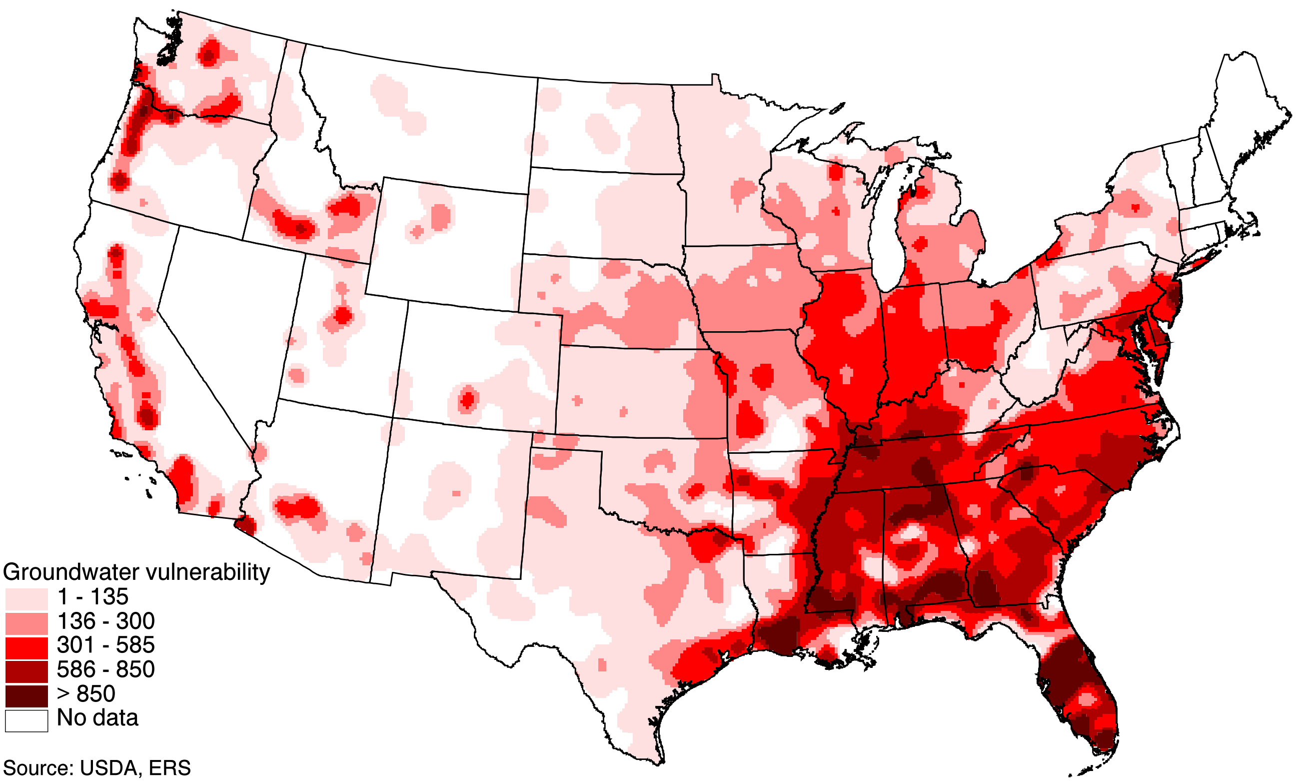
Data developed by the Economic Research Service of the USDA were used to identify regions most vulnerable to nitrate problems (see box 1.3). (Data are not yet available to conduct a similar analysis for phosphorus). Residual nitrogen on cropland (nitrogen from commercial fertilizer, manure, and natural sources in excess of plant needs) is an indicator of potential nitrate availability for runoff to surface water or leaching to ground water. Regions with relatively high residual nitrogen include the Corn Belt, parts of the Southeast, and the intensively irrigated areas of the West (fig. 1.1). Whether residual nitrogen actually contaminates water depends on the leaching characteristics of the soil and on precipitation. For example, regions with the greatest potential for nitrate contamination of groundwater mainly include parts of the Lower Mississippi River and the Southeast, based on an index of groundwater vulnerability that considers factors such as soil type and depth to ground water (Kellogg, Maizell, and Goss, 1992) (fig. 1.2). A simi-

Figure 1.1
**Residual nitrogen, including manure**




Source: USDA, ERS

Figure 1.2
**Groundwater vulnerability index for nitrogen, including manure**




Source: USDA, ERS

## Box 1.3—Using GIS To Create the Maps

**Residual Nitrogen Including Manure** (fig. 1.1). Residual nitrogen is that portion of nitrogen available from natural and manmade sources that is not taken up by crops. Residual nitrogen on cropland (nitrogen from both commercial and manure sources in excess of plant needs) is an indicator of potential nitrate availability for runoff to surface water or leaching to ground water. Data for this figure include commercial fertilizer applications and manure use by farmers recorded in ERS/NASS Cropping Practices, Area Studies, Fruit, and Vegetable Surveys during 1990-1993 (USDA-ERS/NASS). Manure application rates were calculated from 1992 Census of Agriculture data on livestock numbers and average livestock densities by animal type. Nitrogen fixation by legumes in the rotation and nitrogen uptake by crops were estimated using standard agronomic coefficients (Meisinger, 1984; Meisinger and Randall, 1991).

**Groundwater Vulnerability Index for Nitrogen, Including Manure** (fig. 1.2). Nitrate leaching depends on the quantity of residual nitrogen above crop needs and the leachability of the soils to which it is applied. Residual nitrogen, calculated as above, is combined with the leaching characteristics of the soil and the rainfall characteristics in an index of vulnerability to leaching (Kellogg and others, 1992).

**Manure Nitrogen per Acre of Onsite Cropland, 1992** (fig. 1.3). The amount of nitrogen from manure per acre of land available to the operation for land disposal is an indicator of potential problems with excessive manure nitrogen. Economically recoverable nitrogen in manure from confined cattle, swine, and poultry per acre of cropland and managed pasture on the operation is a more sensitive measure than the ratio of nitrogen from manure to total cropland because livestock operators may not have access to much of the land in a county. This measure was developed by Letson and Gollehon from census farm micro data at the U.S. Bureau of the Census, accounting for disclosure restrictions (Letson and Gollehon, 1996).

**Nitrogen From Point Sources** (fig. 1.4). National Pollution Discharge Elimination System (NPDES) permit data on nitrogen discharged by point sources is reported by EPA in the Permit Compliance System (PCS). Both municipal sewage treatment plants and industrial point sources are required to have NPDES permits (Moreau, 1994). However, because ambient pollutant levels may not be nitrogen-limited, or because nitrogen reductions may not otherwise be called for, many point sources that could be expected to have nitrogen discharges report none.

**Groundwater Vulnerability Index for Pesticides, Weighted by Persistence and Toxicity** (fig. 1.5). The amount of active pesticide ingredient applied is an inadequate measure of ground water vulnerability because it does not account for the time the pesticide remains in contact with the environment, the relative seriousness of exposure, and the likelihood that the pesticide will be leached. Data for this figure include pounds of active ingredients in pesticide applications by farmers recorded in ERS/NASS Cropping Practices, Area Studies, Fruit and Vegetable Surveys during 1990-1993 (USDA-ERS/NASS). Persistence of the material in the environment is proportional to the half-life of the material (Kellogg, Maizel, and Goss, 1992). The seriousness of exposure is inversely proportional to the toxicity of the material, measured by the lethal dose (LD50) in rats. Pesticide leaching depends on the characteristics of the active ingredient with regard to solubility and transport, and the leachability of the soils to which it is applied. Pesticide characteristics are combined with the leaching characteristics of the soil and the rainfall characteristics in an index of vulnerability to leaching (Kellogg and others, 1992).

lar index is not yet available for surface water. However, areas with high residual nitrogen and low soil permeability would tend to have a high surface-water vulnerability.

Nitrogen from animal waste is an important source of total nitrogen loads in some parts of the country. A USGS study of nitrogen loadings in 16 watersheds found that manure was the largest source in 6, primarily in the Southeast and Mid-Atlantic States (Puckett, 1994).

Nitrogen (and other contaminants) from manure is an increasing concern given the recent trend toward larger, more specialized beef, swine, and poultry operations. Approximately 450,000 operations nationwide confine or concentrate animals (EPA, 1998a). Of these, about 6,600 have more than 1,000 animal units,

## Box 1.4—Animal Waste Storage Failures

The growing concerns over concentrated animal operations were highlighted in June 1996 when a dike surrounding a large hog-waste lagoon in North Carolina failed, releasing an estimated 25 million gallons of hog waste (twice the volume of the oil spilled by the Exxon Valdez) into nearby fields, streams, and the New River (Satchell, 1996). The 8-acre earthen lagoon was built to allow microbes to digest the waste, and is a common form of management for confined operations. The spill killed virtually all aquatic life in the 17-mile stretch between Richlands and Jacksonville, NC (Satchell, 1996).

There are approximately 6,000 confined animal operations with at least 1,000 animal units in the United States (Letson and Gollehon,1996). (One animal unit equals 1 beef head, 0.7 dairy head, 2.5 hogs, 18 turkeys, or 100 chickens.) Under the Clean Water Act, these facilities cannot discharge to waters except in the event of a 25-year/24-hour storm. This requirement necessitates the construction of onsite storage facilities for holding manure and runoff. In addition to these large operations, facilities with more than 300 animal units that discharge directly to waters are required to take the same measures. Regions with large numbers of animal operations containing more than 1,000 animal units include the Northern Plains (for beef), Pacific (dairy), Corn Belt (swine), Appalachian (swine), and Southeast (broilers) (Letson and Gollehon, 1996).

Most States are responsible for carrying out Clean Water Act regulations. A survey of livestock waste control programs in 10 Midwest and Western States indicated that few States actively inspect facilities for problems, including the integrity of storage structures (Iowa Dept. Nat. Res., 1990). National estimates of broken or leaking storage facilities do not exist. However, a North Carolina State University study estimated that wastes were leaking from half of North Carolina's lagoons built before 1993 (Satchell, 1996), so the problem may be widespread.

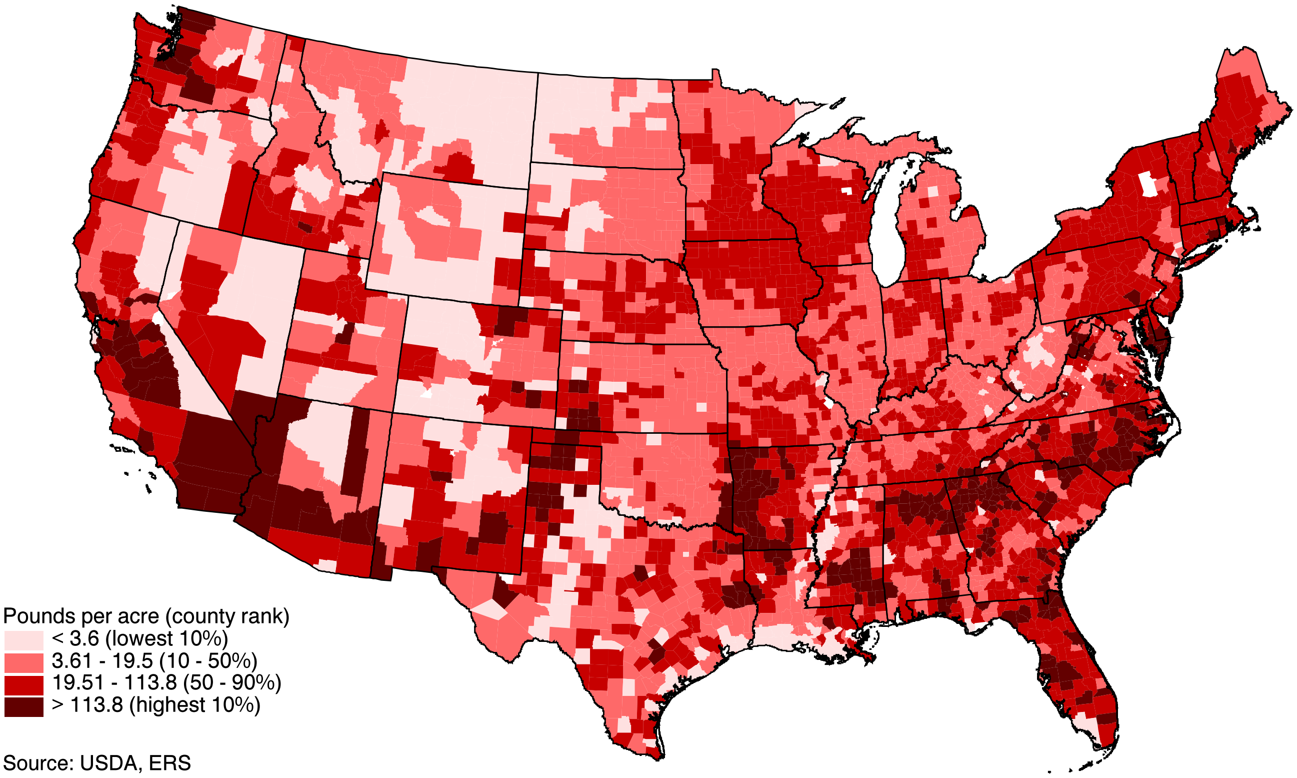
and are defined under the Clean Water Act as Concentrated Animal Feeding Operations, or CAFO's. Such operations must handle large amounts of animal waste, and can cause two sources of water quality problems. First, CAFO's require large and sophisticated manure handling and storage systems, which have at times failed with serious local consequences (see box 1.4, "Animal Waste Storage Failures"). Second, CAFO's tend to lack sufficient cropland on which manure can be spread without exceeding the plants' nutrient needs (Letson and Gollehon, 1996). The highest ratios of manure nitrogen to land are mostly found in parts of the Southeast, Delta, and Southwest (fig. 1.3).

Agricultural activities are not the only source of nutrient pollution. Other loadings stem from point sources such as wastewater treatment plants, industrial plants, and septic tanks. Atmospheric deposition of pollutants is another nonpoint source of nitrogen. Indeed, more than half the nitrogen emitted into the atmosphere from fossil fuel-burning plants, vehicles, and other sources is deposited on U.S. watersheds (Puckett, 1994). The shares of total nitrogen load to selected eastern U.S. estuaries from atmospheric deposition have been estimated to range between 4 and 80 percent (Valigura, Luke, Artz, and Hicks, 1996).

The shares of point and nonpoint sources vary by region, with commercial agricultural fertilizers the dominant source in some areas of the West, and in the central and southeastern United States (Puckett, 1994). Nitrogen discharges from point sources such as sewage treatment plants and fertilizer plants, based on National Pollution Discharge Elimination System permits, are concentrated in the Northeast and Lake States, often areas with major population centers and large concentrations of industry (fig. 1.4). By comparing figures 1.1 and 1.4, one can identify regions where water resources are likely stressed by both point and nonpoint sources of nitrogen. These include the eastern Corn Belt, Florida, Mid-Atlantic, and the agricultural valleys of California.

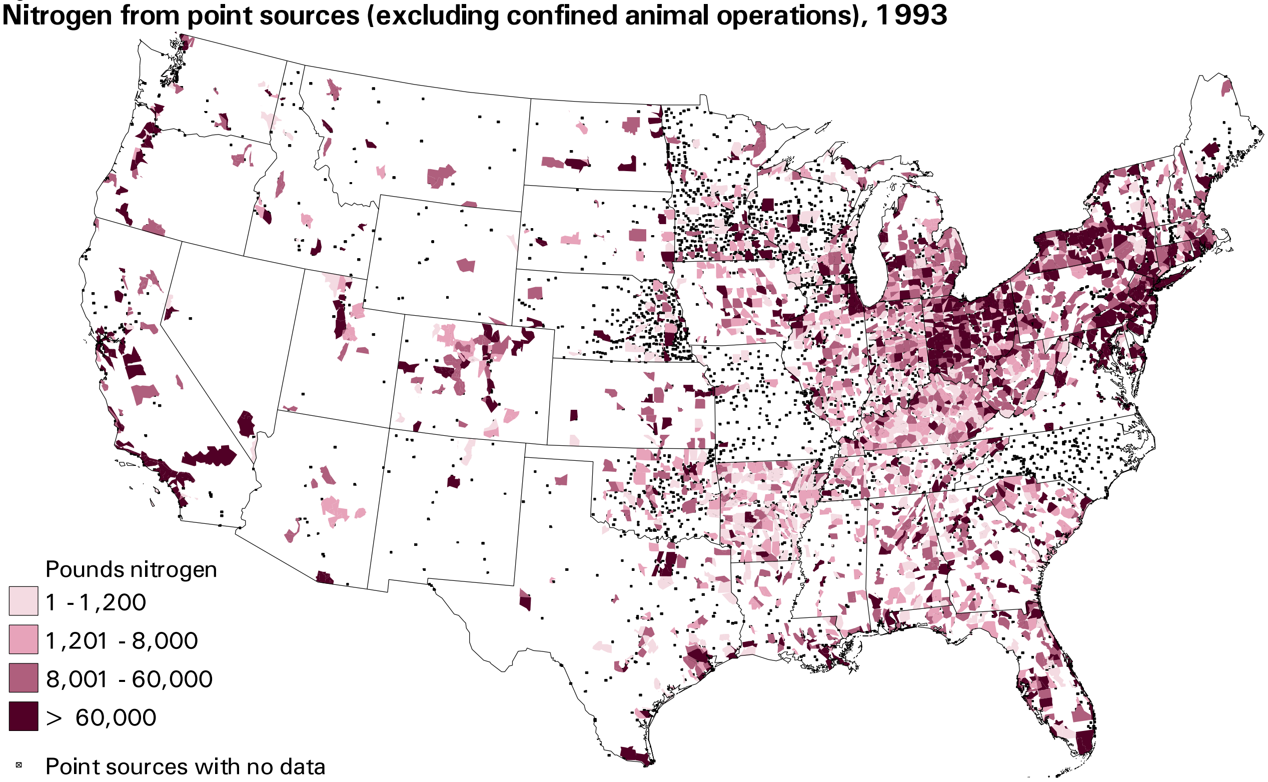
The cost of nutrients in water resources has not been fully estimated. EPA (1997a) estimated costs of $200 million for additional drinking water treatment facilities to meet Federal nitrate standards. Also, consumers are estimated to be willing to pay significant sums to reduce nitrate in the water. Crutchfield, Feather, and Hellerstein (1995) estimated total consumer willingness to pay for reduced nitrate in drinking water in four areas of the United States to be about $350 million per year.

Figure 1.3
**Manure nitrogen per acre of onsite cropland, 1992**




Source: USDA, ERS

Figure 1.4
**Nitrogen from point sources (excluding confined animal operations), 1993**




Source: USDA, ERS, based on EPA permit compliance system data.

## Pesticides

A wide variety of pesticides are applied to agricultural crops to control insect pests, fungus, and disease. Well over 500 million pounds (active ingredient) of pesticides are applied annually on farmland, and certain chemicals can travel far from where they are applied (Smith, Alexander, and Lanfear, 1993; Goolsby and others, 1993). Pesticides move to water resources much as nutrients do, in runoff, run-in, and leachate. In addition, pesticides can be carried into the air attached to soil particles or as an aerosol, and deposited into water bodies with rainfall. Which route a pesticide takes depends on its physical properties and the properties of the soil.

Pesticide residues reaching surface-water systems may harm freshwater and marine organisms, damaging recreational and commercial fisheries (Pait, DeSouza, and Farrow, 1992). Pimentel and others (1991) estimate that direct annual losses from fish kills due to pesticides are less than $1 million, though the authors considered their result an underestimate.

Pesticides in drinking water supplies may also pose risks to human health. Some commonly used pesticides are probable or possible human carcinogens (Engler, 1993). Regulation requires additional treatment by public water systems when certain pesticides exceed health-safety levels in drinking water supplies, though exemptions are specified (42 U.S.C. §300). Enforceable drinking water standards have been established for 15 currently used pesticides, and more are pending (see box 1.5, "Maximum Contaminant Levels"). EPA (1997a) estimates that costs for additional treatment facilities needed to meet current regulations for pesticides and other specific chemicals would be about $400 million, with about another $100 million required over the next 20 years.

Pesticides are commonly detected in water quality studies, though usually at low levels. USGS (1997) detected at least one pesticide in every sampled stream and in about half of sampled ground water in 20 major U.S. watersheds. Pesticides in water supplies have been scrutinized in the Midwest, where large amounts of pesticides are used. Goolsby and others (1993) found that herbicides are detected throughout the year in the rivers of the Midwest, including the Mississippi River. Concentrations are highest during the spring when most pesticides are applied and when spring rains occur. The amounts transported by streams and rivers in the Midwest are generally less than 3 percent of the amount applied, but can still result in concentrations above the MCL (Goolsby and others, 1993). Atrazine (and its metabolites), alachlor, cyanazine, and metolachlor, used principally for weed control in corn and soybeans, were the principal contaminants detected, and are also the most widely used pesticides in the region. Such chemicals, once in drinking water supplies, are not controlled by conventional treatment technologies (Miltner and others, 1989).

Pesticides may pose a special problem for reservoirs. Results from a study of herbicides in 76 midwestern reservoirs showed that some herbicides are detected more frequently throughout the year in reservoirs than in streams, and except for the spring, at higher concentrations (Goolsby and others, 1993). Many of these reservoirs receive much of their storage during the spring and early summer rains, when runoff from cropland contains high concentrations of herbicides.

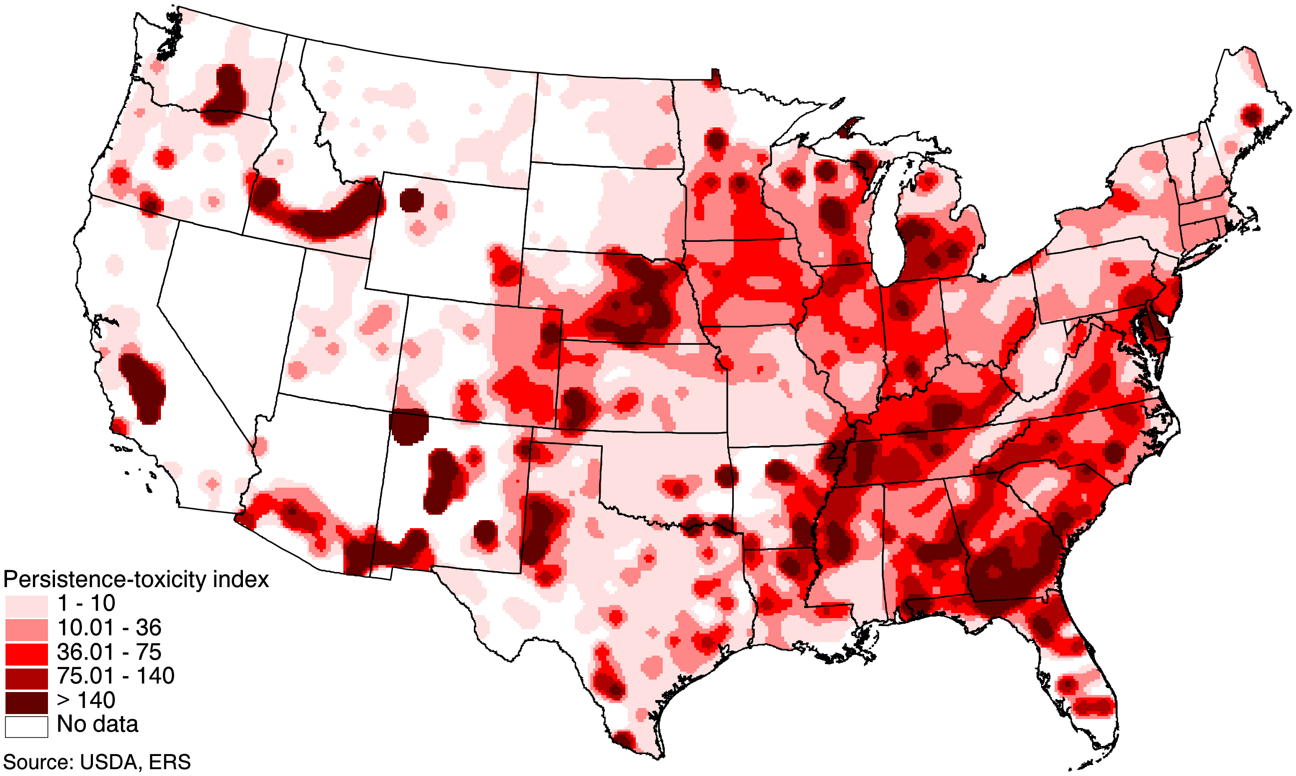
### Box 1.5—Maximum Contaminant Levels

Public water systems are required to ensure that chemicals in the water are below specified thresholds, the maximum contaminant level (MCL) for each chemical. These are enforceable standards, set by EPA, that are considered feasible and safe. MCL's have been set for 15 agricultural chemicals.

| Chemical | MCL (mg/l) | Type chemical |
|---|---|---|
| Nitrate | 10.0 | fertilizer |
| Alachlor | .002 | herbicide |
| Atrazine | .003 | herbicide |
| Carbofuran | .04 | insecticide |
| 2,4-D | .07 | herbicide |
| Dalapon | .2 | herbicide |
| Dinoseb | .007 | herbicide |
| Diquat | .02 | herbicide |
| Endothall | .1 | growth regulator |
| Glyphosate | .7 | herbicide |
| Lindane | .0002 | insecticide |
| Methoxychlor | .04 | insecticide |
| Oxamyl | .2 | insecticide |
| Picloram | .5 | herbicide |
| Simazine | .004 | herbicide |

Source: EPA, 1994a.

Figure 1.5
**Groundwater vulnerability index for pesticides weighted by persistence and toxicity**




Source: USDA, ERS

Because the half-lives of many herbicides are longer in the water than in the soil, relatively high concentrations can persist in reservoirs long after the materials have been applied.

Some pesticides leach into underlying aquifers. Pesticides or their transformation products have been detected in the ground water of 43 States (Barbash and Resek, 1995). EPA's survey of drinking water wells found that 10 percent of the CWS's and 4 percent of rural domestic wells contained at least one pesticide (1992a). However, the EPA estimated that less than 1 percent of the CWS's and rural domestic wells had concentrations above MCL's or Lifetime Health Advisory Levels (the maximum concentration of a water contaminant that may be consumed safely over an average lifetime). In a 1991 study of herbicides and some of their metabolites in near-surface aquifers in the midcontinental United States, USGS detected at least one herbicide in 28.7 percent of the wells sampled (Kolpin, Burkart, and Thurman, 1994). However, no herbicides were found at concentrations greater than the MCL or Lifetime Health Advisory Level. Atrazine and its metabolite desethylatrazine were the most frequently detected compounds.

Groundwater vulnerability to pesticides varies geographically, depending on soil characteristics, pesticide application rates, and the persistence and toxicity of the pesticides used (fig. 1.5) (see box 1.3 for a description of how the map was created). Areas with sandy, highly leachable soils and high application rates of toxic or persistent pesticides, such as central Nebraska, generally have high vulnerability ratings. Irrigated areas in Idaho, California, Texas, Washington, and the Southeast also have high vulnerability ratings. Despite widespread use of pesticides, the Corn Belt ranks lower than some of the above-mentioned areas because the predominant soils are not prone to leaching, are not irrigated, or because the chemicals used (mostly herbicides) are less persistent or toxic.

## Salts

When irrigation water is applied to cropland, a portion of it runs off the field into ditches and flows back to a receiving body of water. These irrigation return flows may carry dissolved salts, as well as nutrients and pesticides, into surface or ground water. Increased concentrations of naturally occurring toxic minerals, such as selenium and boron, can harm aquatic wildlife and degrade recreational opportunities. Increased levels of dissolved solids in public drinking water can increase water treatment costs, force the development of alternative water supplies, and reduce the lifespans of water-using household appliances. Increased salinity levels in irrigation water can reduce crop yields or damage soils so that some crops can no longer be grown.

Dissolved salts and other minerals are an important cause of pollution in the Southern Plains, arid Southwest, and southern California. Total damages from salinity in the Colorado River range from $310 million to $831 million annually, based on the 1976-85 average levels of river salinity. These include damages to agriculture ($113-$122 million), households ($156-$638 million), utilities ($32 million), and industry ($6-$15 million) (Lohman, Milliken, and Dorn, 1988).

The USGS reports mixed trends of salinity in surface water over the 1980's (Smith, Alexander, and Lanfear, 1993). Measures of dissolved solids (mostly ions of calcium, magnesium, sodium, potassium, bicarbonate, sulfate, and chloride) indicate that water quality improved at more stations than it worsened. Salinity trends in water for domestic and industrial purposes generally improved during the 1980's, though salinity worsened for irrigation purposes. Among USGS cataloguing units (watersheds) with significant irrigation surface-water withdrawals, the share with annual average dissolved solids concentrations greater than 500 mg/L increased from 30 percent in 1980 to 33 percent in 1989 (Smith, Alexander, and Lanfear, 1993).

### Pathogens

The possibility of pathogen-contaminated water supplies is attracting increased attention (NRAES, 1996; Olson, 1995). Bacteria are the third leading source of impairment of rivers and the second leading cause in estuaries (EPA, 1998a). Potential sources include inadequately treated human waste, wildlife, and animal operations. Animal waste contains pathogens that pose threats to human health (CAST, 1996). Microorganisms in livestock waste can cause several diseases through direct contact with contaminated water, consumption of contaminated drinking water, or consumption of contaminated shellfish. Bacterial, rickettsial, viral, fungal, and parasitic diseases are potentially transmissible from livestock to humans (CAST, 1996). Fortunately, proper animal management practices and water treatment minimize the risk to human health posed by most of these pathogens. However, protozoan parasites, especially *Cryptosporidium* and *Giardia*, are important etiologic agents of water-borne disease outbreaks (CDC, 1996). *Cryptosporidium* and *Giardia* may cause gastrointestinal illness, and *Cryptosporidium* may lead to death in immunocompromised persons. These parasites have been commonly found in beef herds, and *Cryptosporidium* is estimated to be prevalent on dairy operations (USDA, APHIS, 1994; Juranek, 1995).

Outbreaks of waterborne diseases are a growing concern. EPA (1997a) estimates the cost of facilities for improved microbial treatment to be about $20 billion over the next 20 years, with about half of that needed immediately. The health cost of *Giardia* alone is estimated to be $1.2-$1.5 billion per year (EPA, 1997b). Cryptosporidia is a more recently identified threat, with oocysts present in 65-97 percent of surface water sampled in the United States (CDC, 1995). The organism has been implicated in gastroenteritis outbreaks in Milwaukee, Wisconsin (400,000 cases and 100 deaths in 1993) and in Carrollton, Georgia (13,000 cases in 1987). The cost of the Milwaukee outbreak is estimated to exceed $54 million (*Health and Environment Digest*, 1994). While the source of the organism in these outbreaks was never determined, its occurrence in livestock herds has brought some attention to this sector, especially given the proximity of cattle and slaughterhouses to Milwaukee (MacKenzie and others, 1994).

## Costs of Pollution

The total costs of water pollution from point and nonpoint sources are largely unknown. Research has examined the costs of some specific pollutants (e.g., sediment) or the costs of poor water on some desired uses (e.g., recreation). Other indicators of damages include the estimated benefits from pollution control efforts, which give a lower bound to damages (table 1-3). Water quality damages due to sediment from soil erosion are substantial, and appear greater than estimated damages from other pollutants (nutrients, pesticides, and pathogens).

**Table 1-3—National estimates of the damages from water pollution or benefits of water pollution control**

| Estimate of— | Study/year | Description |
|---|---|---|
| | | *Selected estimates of annual damages* |
| Water quality damages from soil erosion | Clark and others (1985) | Damages to all uses: $3.2-$13 billion, “best guess” of $6.1 billion (1980 dollars). Cropland’s share of damages: $2.2 billion. |
| Water quality damages from soil erosion | Ribaudo (1989) | Damages to all uses: $5.1-$17.6 billion, “best guess” of $8.8 billion. Agriculture’s share of damages: $2-$8 billion. |
| Adjustments to net farm income considering effects of soil erosion | Hrubovcak, LeBlanc, and Eakin (1995) | Reduction in net farm income account of about $4 billion due to soil erosion effects. |
| Environmental costs of pesticides | Pimentel and others (1991) | Direct costs from fish kills: less than $1 million. |
| Infrastructure needs to protect drinking water from poor source-water quality | EPA (1997a) | $20 billion in current and future (20-year) need under Safe Drinking Water Act requirements for microbial treatment; $0.2 billion for nitrates; and $0.5 billion for other synthetic chemicals, including pesticides. |
| Health costs from waterborne disease outbreaks | EPA (1997b) | Damages from *Giardia* outbreaks: $1.2-$1.5 billion in health costs. |
| Recreational damages of water pollution | Freeman (1982) | Total recreational damages from all forms of water pollution: $1.8-$8.7 billion; “best guess” of $4.6 billion (1978 dollars/year). |
| | | *Selected estimates of annual benefits from water pollution control* |
| Water quality benefits of reduced soil erosion from conservation practices | Ribaudo (1986) | Erosion reduction from practices adopted under the 1983 soil conservation programs were estimated to produce $340 million in offsite benefits over the lives of the practices. |
| Water quality benefits of reduced soil erosion from Conservation Reserve Prog. | Ribaudo (1989) | Reducing erosion via retirement of 40-45 million acres of highly erodible cropland would generate $3.5-$4.5 billion in surface-water quality benefits over the life of the program. |
| Recreational fishing benefits from controlling water pollution | Russell and Vaughan (1982) | Total benefits of $300-$966 million, depending on the quality of fishery achieved. |
| Recreational benefits of surface-water pollution control | Carson and Mitchell (1993) | Annual household willingness to pay for improved recreational uses of $205-$279 per household per year, or about $29 billion. |
| Recreational benefits of soil erosion reductions | Feather and Hellerstein (1997) | Total of $611 million in benefits from erosion reductions on agricultural lands since 1982, based on recreation survey data. |

# Programs for Controlling Agricultural Pollution

Agricultural nonpoint-source pollution (NPS) is addressed at both Federal and State levels. A host of programs using several types of policy instruments have been implemented.

## Federal Programs

At the Federal level, EPA is chiefly responsible for policies and programs that deal with water quality, mainly under provisions of the Federal Water Pollution Control Act of 1972 (the Clean Water Act). The Act deals with point-source pollution through technology-based controls (uniform, EPA-established standards of treatment that apply to certain industries and municipal sewage treatment facilities), and water quality-based controls that invoke State water quality standards (Moreau, 1994). The National Pollutant Discharge Elimination System (NPDES) sets limits on individual point-source effluents. Large, confined animal operations (over 1,000 animal units) fall under the NPDES, though enforcement has been a problem, and many facilities lack permits (Westenbarger and Letson, 1995).

When technology-based controls are inadequate for waters to meet State water quality standards, Section 303(d) of the Clean Water Act requires States to identify those waters and to develop total maximum daily loads (TMDL) (EPA, 1993). A TMDL is the sum of individual wasteload allocations for point sources, load allocations for nonpoint sources and natural background, and a margin of safety (Graham, 1997). A TMDL approach forces the accounting of all sources of pollution. This helps identify how additional basin reductions, if needed, might be obtained. EPA has responsibility for developing TMDL's if a State fails to act (EPA, 1993). Over 500 TMDL plans have been initiated since 1992, and 225 have been completed and approved by EPA (EPA, 1997).

NPS pollution is dealt with directly in several programs authorized by the Clean Water Act. Section 319 established EPA's Nonpoint Source Program, which grants States funds to develop and promote nonpoint-source management plans and other programs. EPA also provides program guidance and technical support under the program. States had a deadline of 1995 for developing and implementing nonpoint-source management plans. Under the Clean Lakes Program (sec. 314), EPA provides grants to States for various activities, including projects to restore and protect lakes. The National Estuary Program (sec. 320) helps States develop and implement basinwide comprehensive programs to conserve and manage their estuary resources.

The Coastal Zone Management Act Reauthorization Amendments (CZARA) is the first federally mandated program requiring specific measures to deal with agricultural nonpoint sources (16 U.S.C. §§ 1455(d)(16), 1455b). CZARA requires each State with an approved coastal zone management program to submit a program to "implement management measures for NPS pollution to restore and protect coastal waters" (cited in USDA, ERS, 1997). States can first try voluntary incentive mechanisms, but must be able to enforce management measures if voluntary approaches fail. Implementation of plans is not required to begin until 2004.

The Safe Drinking Water Act (SDWA) requires the EPA to set standards for drinking water quality and requirements for water treatment by public water systems (Morandi, 1989). The SDWA authorized the Wellhead Protection Program in 1986 to protect supplies of ground water used as public drinking water from contamination by chemicals and other hazards, including pesticides, nutrients, and other agricultural chemicals (EPA, 1993). The program is based on the concept that land-use controls and other preventive measures can protect ground water. As of December 1998, 45 States had an EPA-approved wellhead protection program (EPA, Office of Ground Water and Drinking Water, 1998). The 1996 amendments to the SDWA require EPA to establish a list of contaminants for consideration in future regulation (EPA, 1998b). The Drinking Water Contaminant Candidate List, released in March 1998, lists chemicals by priority for (a) regulatory determination, (b) research, and (c) occurrence determination. Several agricultural chemicals, including metolachlor, metribuzin, and the triazines, are among those to be considered for potential regulatory action (EPA, 1998b). EPA will select five contaminants from the "regulatory determination priorities" list and determine by August 2001 whether to regulate them to protect drinking water supplies.

Also under the 1996 amendments, water suppliers are required to inform their customers about the level of certain contaminants and associated EPA standards, and the likely source(s) of the contaminants, among other items (EPA, 1997e). If the supplier lacks specific

**Table 1-4—USDA programs associated with water quality and the incentives they employ**

| Program | Economic | Educational | Research & Development |
|---|---|---|---|
| Environmental Quality Incentives | X | X | |
| Water Quality | X | X | X |
| Conservation Technical Assistance | | X | |
| Conservation Compliance | X | X | |
| Conservation Reserve | X | | |
| Wetlands Reserve | X | | |

See USDA, ERS (1997) for a description of these programs.

information on the likely source(s), set language must be used for the contaminants, such as “runoff from herbicide used on row crops” (e.g. for atrazine). “The information contained in the consumer confidence reports can raise consumers’ awareness of where their water comes from,...and educate them about the importance of preventative measures, such as source water protection...” (*Federal Register*, August 19. 1998, p. 44512). Increased consumer awareness concerning water supplies could lead to public pressure on farmers to reduce pesticide use (Smith and Ribaudo, 1998).

USDA administers a variety of water quality programs that directly involve agricultural producers (table 1-4). These programs use financial, educational, and research and development tools to help improve water quality and achieve other environmental objectives. The Environmental Quality Incentives Program (EQIP), authorized by the Federal Agriculture Improvement and Reform Act of 1996, provides technical, educational, and financial assistance to eligible farmers and ranchers to address soil, water, and related natural resource concerns on their lands in an environmentally beneficial and cost-effective manner (USDA, NRCS, 1998). This program consolidated the functions of a number of USDA programs, including the Agricultural Conservation Program, Water Quality Incentives Program, Great Plains Conservation Program, and Colorado River Basin Salinity Program. EQIP assistance is targeted to priority conservation areas and identified problems outside of those areas. Five- to 10-year contracts may include incentive payments as well as cost-sharing of up to 75 percent of the costs of installing approved practices. Fifty percent of the program funding is to be targeted at natural resource concerns related to livestock production (USDA, NRCS, 1998). Owners of large, concentrated livestock operations are not eligible for cost-share assistance for installing animal waste storage or treatment facilities. However, technical, educational, and financial assistance may be provided for other conservation practices on these large operations. EQIP is designed to maximize environmental benefits per dollar expended (USDA, NRCS, 1998).

The Water Quality Program (WQP), established in 1990 and essentially completed, has attempted to determine the precise nature of the relationship between agricultural activities and water quality. It has also attempted to develop and induce adoption of technically and economically effective agrichemical management and production strategies that protect surface- and groundwater quality (USDA, 1993). WQP includes three main components: (1) research and development; (2) education, technical, and financial assistance; and (3) database development and evaluation. The first two components were carried out in targeted project areas. Seven projects were devoted to research and development (Management System Evaluation Areas) and 242 to assisting farmers implement practices to enhance water quality (Hydrologic Unit Area projects, Water Quality Incentive projects, Water Quality Special projects, and Demonstration Projects). The database development activity consists of annual surveys of chemical use on major field, vegetable, and fruit crops.

Since 1936, USDA has provided technical assistance to farmers for planning and implementing soil and water conservation and water quality practices through Conservation Technical Assistance (CTA) (USDA, ERS, 1997). Farmers adopting practices under USDA conservation programs and other producers who request aid in adopting approved USDA practices are eligible for technical assistance. Some programs have required technical assistance as a condition for receiving financial assistance.

Conservation Compliance provisions were enacted in the Food Security Act of 1985 to reduce soil erosion (USDA, ERS, 1997). Producers who farm highly erodible land (HEL) were required to implement a soil conservation plan to remain eligible for other specified USDA programs that provide financial payments to producers.

Water quality would also be expected to improve from two USDA land retirement programs. The Conservation Reserve Program (CRP) was established in 1985 as a voluntary long-term cropland retirement program (USDA, ERS, 1997). USDA provides CRP participants with an annual per-acre rent and half the cost of establishing a permanent land cover in exchange for retiring highly erodible or other environmentally sensitive cropland for 10-15 years. U.S. cropland erosion has been reduced by about 20 percent under the program (USDA, ERS, 1994). The Wetlands Reserve Program, authorized as part of the Food, Agriculture, Conservation, and Trade Act of 1990, is primarily a habitat protection program, but retiring cropland and converting back to wetlands also has water quality benefits (USDA, ERS, 1997). These benefits include not only reduced chemical use and erosion on former cropland, but also the ability of the wetland to filter sediment and agricultural chemicals from runoff and to stabilize stream banks.

In addition to the above programs that provide direct assistance to producers, USDA also provides assistance to State agencies and local governments through the Small Watershed Program (otherwise known as Public Law 566) (USDA, ERS, 1994). To help prevent floods, protect watersheds, and manage water resources, this program includes establishment of measures to reduce erosion, sedimentation, and runoff.

### State Programs

Most, if not all, States provide incentives to farmers to adopt management practices that reduce agricultural NPS pollution. Common strategies include watershed and land-use planning, development of voluntary best management practices, technical assistance programs, and cost-sharing for prevention and control measures.

Recently, more States have been moving beyond a voluntary approach to address NPS pollution. Mechanisms to enforce certain behavior include regulation and liability provisions (ELI, 1997). State laws using such provisions for NPS pollution vary widely in definitions, enforcement mechanisms, scope, and procedures, largely because of the absence of Federal direction (ELI, 1997). Catalysts moving States toward stronger measures include immediate and urgent problems (such as nitrate contamination of ground water in Nebraska, animal waste problems in North Carolina, and pesticide contamination of ground water in California and Wisconsin), the use of total maximum daily load provisions for identifying sources of water contaminants, the requirements of the Coastal Zone Act Reauthorization Amendments, and the improving technical ability of States to assess their waters (ELI, 1997).

States are using five different mechanisms to make adoption of best management practices (BMP's) more enforceable (ELI, 1997). These include making BMP's directly enforceable in connection with required plans and permits; making BMP's enforceable if the producer is designated a "bad actor"; making compliance with BMP's a defense to a regulatory violation; making BMP's the basis for an exemption from a regulatory program; and making compliance with BMP's a defense to nuisance or liability actions.

While many States have provisions that deal with water quality as it relates to agricultural NPS pollution, they often target only a subset of water quality problems. Few States deal with agricultural NPS pollution in a comprehensive manner (table 1-5). Most target individual pollutants (sediment), resources (ground water), regions (coastal zone), or type of operations (swine). Most of these laws have been enacted within the past 5 years, so the impacts of these policies on producers have yet to be seen.

## Summary and Policy Implications

Nonpoint sources of pollution are the largest remaining sources of water quality impairment in the United States. While most of the sampled waters are reported to be supporting designated uses, runoff from agriculture, forests, urban areas, and other land uses are causing impairments in some important water resources. Nutrients, bacteria, and siltation are reported to be the largest causes of impairment to surface waters; agriculture is the primary source of impairments in rivers and lakes, and a major source in estuaries (EPA, 1998a). Both Federal and State governments have responded with primarily voluntary programs for addressing nonpoint-source pollution, though some States are moving toward stronger policy measures.

Deficiencies in water quality data hinder the development of a full range of water quality policies at Federal and State levels, and complicate measuring the progress of initiatives already undertaken. Data are often unable to identify the relative contributions of pollutant loadings from different sources (Knopman and Smith, 1993). In many cases, current monitoring efforts of nonpoint-source pollution are incapable of attributing changes in water quality to the actions of a specific polluter. How these deficiencies affect policy development is addressed more fully in the next chapter.

**Table 1-5—Summary of State water quality mechanisms for controlling agricultural pollution[1]**

| State | Nutrient plan requirement | Pesticide restriction | Sediment restriction | Animal waste disposal plan | Comprehensive |
|---|---|---|---|---|---|
| Alabama | | | | | |
| Alaska | | | | | |
| Arizona | X | X | | X | |
| Arkansas | | | | X | |
| California | | X | | X | |
| Colorado | X | | | X | |
| Connecticut | | | | X | |
| Delaware | | | | | |
| Florida | X | | | X | |
| Georgia | | | | | |
| Hawaii | | | | | |
| Idaho | | | | | |
| Illinois | | | | X | |
| Indiana | | | | | |
| Iowa | X | X | | X | |
| Kansas | | X | | X | |
| Kentucky | | | | X | X |
| Louisiana | | | | | |
| Maine | | | | X | X |
| Maryland | X | | X | X | X |
| Massachusetts | | | | | |
| Michigan | | | | | |
| Minnesota | | | | X | |
| Mississippi | | | | X | |
| Missouri | | | | X | |
| Montana | X | X | | X | |
| Nebraska | X | | X | X | |
| Nevada | | | | | |
| New Hampshire | | | | | |
| New Jersey | | | | | |
| New Mexico | | | | | |
| New York | | | | | |
| North Carolina | | | | X | |
| North Dakota | | | | | |
| Ohio | | | X | X | |
| Oklahoma | | | | X | |
| Oregon | | | | X | X |
| Pennsylvania | | | X | X | |
| Rhode Island | | | | | |
| South Carolina | | | | X | |
| South Dakota | | | | X | |
| Tennessee | | | | X | |
| Texas | | | | | |
| Utah | | | | | |
| Vermont | | | | X | X |
| Virginia | | | | X | X |
| Washington | | | | X | X |
| West Virginia | X | | | X | |
| Wisconsin | X | X | | X | X |
| Wyoming | | | | X | |

[1] Mechanisms may apply only under certain conditions or in certain localities.
Sources: ELI, 1998; NRDC,1998; Animal Confinement Policy National Task Force, 1998.

## Chapter 2

# Comparing Options for Addressing Nonpoint-Source Pollution

*Water pollution is an externality to production that prevents an efficient allocation of resources. One role of public policy is to correct such externalities. To do so, an agency must take into account a number of considerations in selecting a policy instrument. Weighing on these considerations are the unique characteristics of agricultural nonpoint source pollution. Nonpoint source pollution cannot be easily traced back to individual sources, and its movement is a stochastic process related to weather, topography, and land use. However, limitations in information do not prevent the design of economically sound pollution control policies.*

## Introduction and Overview

In chapter 1, we showed the economic costs associated with nonpoint-source pollution to be significant. In this chapter, we formalize the nonpoint problem by first discussing the characteristics of nonpoint-source pollution and then examining why government intervention is necessary. Next, we focus on how the unique characteristics of nonpoint-source pollution influence policy design and limit the options for cost-effective control. Finally, issues related to the appropriate level of government (Federal, State, local) for carrying out nonpoint-source pollution policies are discussed.

## Characteristics of Nonpoint-Source Pollution

Agricultural water pollution is described as "nonpoint source" (NPS) pollution because emissions (runoff) from each farm are diffuse. Runoff does not emanate from a single point, but leaves each field in so many places that accurate monitoring would be prohibitively expensive (Braden and Segerson, 1993; Shortle and Abler, 1997). The amount and quality of runoff leaving a field depend not only on factors that can be measured, such as the technology used and the use of variable inputs, but also on factors such as rainfall that vary daily and are difficult to predict (Braden and Segerson, 1993; Shortle and Abler, 1997).[1]

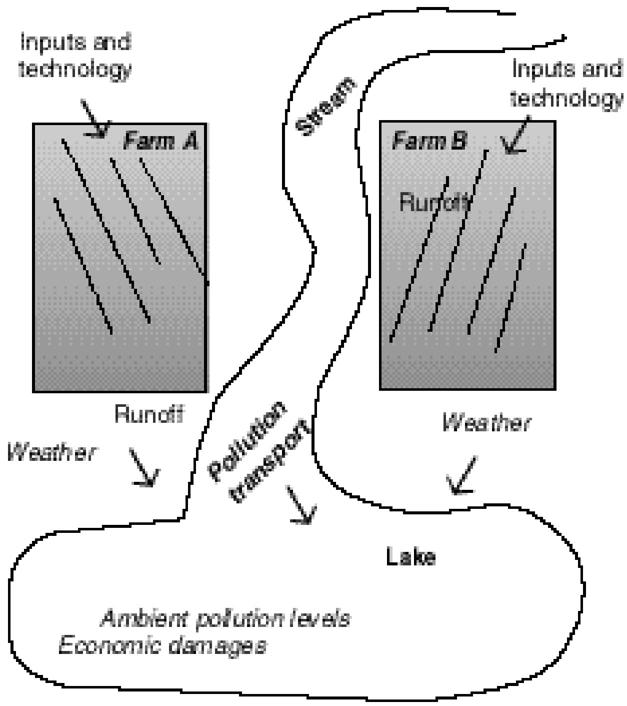
The relationship between agricultural production and damages from water pollution is complex, involving physical, biological, and economic links (fig. 2.1). How well a policy performs often depends on how well these links are understood. The first link (runoff) is between production practices and movement of pollutants off a field. Important variables include rainfall, soil characteristics, slope, crop management, chemical management, water management, and conservation practices. These factors combine to determine the amount of soil particles, nutrients, and pesticides that actually leave a field.

The second link consists of pollutants moving from the field to water resources, or the pollution transport process. Pollutants can travel in overland runoff and be discharged directly into the water resource, or enter small streams and waterways and be transported to larger rivers, lakes, and estuaries. The amount of pollutants that eventually reach a water resource depends on factors such as distance, rainfall, slope, vegetation, properties of agrichemicals, and intervening conservation practices such as riparian buffers and constructed wetlands.

The third link is between the agricultural pollutants discharged into water resources and water quality. Water quality is expressed in terms of physical and biological measures, including dissolved oxygen, temperature, turbidity, pH, ambient pollution concentrations, fish populations, algae levels, and zooplankton and bacterial concentrations. Changes in ambient pol-

[1] Inputs are defined as those items used in production that can be applied in varying amounts (e.g., chemical fertilizers, pesticides, water for irrigation, etc.). Alternatively, technologies (or management practices) are defined as specific production techniques or methods used (e.g., conservation tillage, crop rotation, aerial pesticide applications, etc.).

Figure 2.1
**Flow of agricultural pollution**




lution concentrations may affect other measures of water quality (fish populations) as well.

The fourth link is how changes in ambient pollution levels (and hence water quality) affect the ability of the water resource to provide economic services. For example, the recreation potential for a water body can be affected by changes in its biological characteristics and physical appearance. Fewer fish, foul odors, algae blooms, and turbidity can all reduce the attractiveness of a potential recreation site. Suspended sediment, algae, and dissolved chemicals can increase the cost of providing water for municipal use.

The fifth link is between the services provided by the water resource and the economic (use and nonuse) value actually placed on those services. This is a function of demand by individuals, municipalities, or industry. The greater the demand for services such as recreation or industrial use, the greater their value and the greater the economic damages if impaired by pollution. Factors influencing the value of services include population, regional income, and treatment costs. The reduction in economic values due to ambient pollution levels is referred to as economic damages.

## Nonpoint-Source Pollution Is an Externality

Nonpoint-source pollution (NPS) occurs at inefficiently high levels because farmers, when making their production decisions, have no incentive to consider the costs pollution imposes on others (Baumol and Oates, 1979). Economists refer to such costs as *externalities* because they are external to the production manager's decision framework. A decentralized, competitive economy will not maximize social welfare in the face of agricultural NPS pollution; farmers have no incentives to consider the social costs of pollution when making production decisions. Economic theory suggests several ways to design policies that provide the appropriate incentives for farmers to account for the costs of their pollution.

An *efficient solution* is one that maximizes expected net economic benefits—the private net benefits of production (aggregate farm profits) minus the expected economic cost of pollution.[2] Decisions must be made based on the **expectation** of what damages will be since it is impossible to accurately predict damages due to the varying nature of pollutant runoff and transport. Consequently, the efficient solution is often referred to as the ex ante efficient solution, meaning that it is the expected outcome as opposed to the actual or realized outcome.

### Efficiency Conditions

The economically efficient solution is defined by three conditions (formally developed in Appendix 2A):

(1) **For each input and each site, the marginal net private benefits from the use of the input on the site equal the expected marginal external damages from the use of the input.** In other words, the last unit of the input used in production should provide an equal increase in net private benefits and expected damages. This condition is violated and the Pareto-efficient outcome forgone if farmers ignore external damages. Instead, the use of pollution-causing inputs will be too high, the use of pollution-mitigating inputs will be too low, and the resulting runoff levels will be too high.

[2] Private net benefits from production may also include benefits to consumers and owners of factors of production. We discuss private net benefits in terms of aggregate profits for simplicity, but note that nonpoint policies may also impact consumers and factor owners by altering input and output prices.

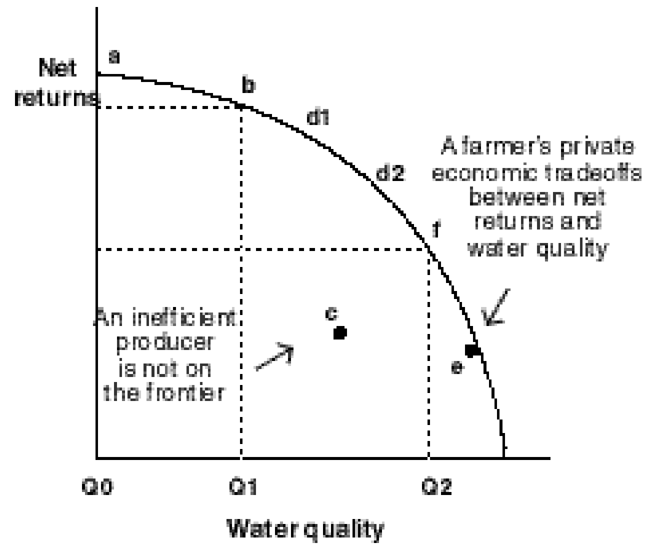
(2) **A site should be brought into production as long as profits on this site are larger than the resulting expected increase in external damage.** In other words, the benefits from allowing a site into production should exceed the expected social costs of doing so. This condition defines the optimal amount of land in production. *Marginal acreage* is defined as sites with profits equal to their expected contribution to damages in the efficient solution (or the sites with the smallest positive difference between profits and expected damage contribution). Sites with a positive (negative) difference between profits and its expected damage contribution are defined as *inframarginal (extra-marginal)*. It is only efficient for the marginal and inframarginal sites to be in production. If external damages are ignored, the amount of land expected to produce profitably is greater than optimal, as is runoff and ambient pollution.

(3) **Technologies should be adopted on each site such that the incremental impact of each technology (relative to the next best alternative) on expected social net benefits is greater than or equal to the incremental impact on expected damages.**

The three efficiency conditions represent economic tradeoffs involving farm profitability (net returns) and water quality (fig. 2-2). Movement along the curve represents changes in inputs and technologies familiar to the farmer that achieve increasing levels of water quality. For instance, higher levels of water quality protection may necessitate a move away from conventional practices to ones using fewer chemical inputs, adding filter strips, and even retiring cropland. It is assumed here that higher levels of water quality can be achieved only with a loss of net returns, reflecting the fact that pollution control is typically costly.

We assume here and throughout that a farmer's economic goal is to maximize net revenues, taking into consideration personal and family health. Suppose tradeoffs exist as in figure 2.2 and that the socially desirable level of water quality is *Q2*. With no external incentives to control pollution and no apparent personal health impacts from the pollution, a farmer's economic calculations would lead to production at point *a*. Point *a* maximizes net returns without consideration of water quality. Any movement away from *a* results in a profit loss. A farmer may have an incentive to pollute less if directly affected by onfarm practices, such as polluting a drinking water well. Such

Figure 2.2
**Farmlevel tradeoff between net returns and water quality, given known technology**




consideration, without any further incentives, may lead to the adoption of practices at point *b*, which corresponds to a water quality level of *Q1*. Essentially, the policy tools discussed in the following chapters aim to move farmers along the frontier toward *Q2*.

## Nonpoint-Source Policy Goals: Cost-Effectiveness[3]

Environmental policies are cost-effective if they achieve some measurable objectives or goals at least cost. An overall strategy for water quality protection, therefore, depends on the choice of both policy goals and the instruments to achieve them. These choices are generally interdependent. Depending on the goals,

[3] While we do not discuss this explicitly, existing market distortions that are outside of the regulatory agency's control must be taken into account when designing optimal incentives. Otherwise, the performance of incentives will be limited. A variety of agricultural policies, such as price floors, target prices, and deficiency payments, that are designed to support farm income also stimulate production. The resulting use of more chemical inputs and more intensive land use may lead to increases in nonpoint-source pollution (Miranowski, 1978; Reichelderfer, 1990; Ribaudo and Shoemaker, 1995). The 1996 Farm Act has phased out many of these policies, explicitly to reduce market distortions. Other programs, such as acreage retirement programs and paid land diversion, are supply control programs that may help to offset the effects of some support policies. Recently, some supply control programs and other agricultural conservation programs (e.g., Sodbuster, Swampbuster) have been targeted to environmentally sensitive land and linked to agricultural support policies.

it may not be possible to attain the least-cost solution with some types of policy instruments.

Water quality protection is costly to those who must pay for pollution reduction. Consequently, nonpoint policies can produce net social economic gains only if their impact is to reduce the expected damages from pollution. Reducing expected damages may not always constitute a measurable policy goal, however, because damages from NPS pollution often remain largely unquantified. In addition, the relationships among runoff, ambient pollution levels, and economic damages to society are often unknown or poorly understood (Shortle, Horan, and Abler, 1998; Baumol and Oates, 1988). Instead, it is necessary to adopt alternative goals that are measurable and that are believed to reduce expected damages—even when damages remain unknown. Potential alternatives include goals based on measurable physical (i.e., ambient water quality, expected runoff) or production-related (i.e., input use, technology) performance indicators.[4] For example, U.S. point-source policy goals are often defined in terms of ambient water quality (EPA, 1993). With nonpoint sources, ambient water quality or runoff goals must be defined in terms of a probability of occurrence (e.g., to attain a mean ambient water quality at least cost) because a particular policy could produce a variety of results due to the natural variability associated with the nonpoint process (Braden and Segerson, 1993; Shortle and Abler, 1997; Shortle, 1990).

There may be instances where achieving certain types of policy goals cost-effectively may increase expected damages. For example, suppose a policy goal is to meet a mean ambient pollution target and that Method A meets this goal at least cost (Method A is the cost-effective approach). Even if mean ambient levels are decreased, Method A may unintentionally increase the variability of pollution levels, increasing expected damages and making society worse off (Shortle, 1990). However, without the ability to measure damages, it may not be possible to recognize when such situations arise.

There are situations where a policy goal is expected to reduce damages, even though damages remain unknown. For example, Method A will reduce mean damages if ambient pollution levels are reduced for each potential state of nature (for each possible realization of random events). Similar results do not apply to runoff-based goals, however, and appropriately specified goals based on ambient pollution levels will generally be preferred (Horan, 1998).

Another problem with physically based policy goals is comparing different methods of achieving the same goal. The economically preferred method of pollution control achieves a goal with greatest expected social net benefits (defined as the sum of private pollution control costs plus the expected benefits of pollution reduction). The economically preferred method is economically superior to all other methods that achieve the same goal because it takes damages into consideration. The other methods, including cost-effective ones, do not. Since damages often remain unquantified, it would be convenient if *economically preferred* and cost-effective methods always coincided. However, the economically preferred method of achieving a physically based goal will generally differ from the cost-effective method that achieves the same goal (Horan, 1998). The differences are due to risk effects that arise because the cost-effective method does not account for the impact of each production choice on expected damages. For example, suppose Policy A corresponds to $50 in control costs and yields a $100 expected reduction in damages, for a net expected social gain of $50. Suppose Policy B achieves the same physical goal at a cost of $60, but the policy yields a $120 expected reduction in damages (a greater reduction in damages may result under Policy B if Policy B reduces the variability of pollution relative to Policy A), for an expected net social gain of $60. Policy A is the cost-effective policy because it achieved the policy goal at a lower cost. Policy B is the economically preferred policy because it generates a greater net social gain, and is the one that policymakers would choose if they had information about damages. However, Policy A will generally be chosen because economic measures of damages are seldom known.[5]

[4] Expected runoff levels could be measured with a simulation model.

[5] With a deterministic pollution process (as is often assumed for point sources of pollution), ambient water quality goals or runoff goals can always be used to ensure a reduction in damages even when damages remain unknown, and to ensure that the least-cost method of achieving particular goals is economically preferred over all other methods. This is because ambient pollution or runoff levels can be controlled with certainty, and a deterministic reduction in these measures would correspond to a reduction in damages. Similar results do not apply to nonpoint pollution control, however,

The final class of goals is based on input use and technology choices. For example, instead of designing policies to reduce mean nitrogen loadings, the goal may be a specified reduction in nitrogen fertilizer application rates (e.g., a 20-percent reduction in nitrogen use in a watershed). Such goals provide policymakers with more direct control (than water quality goals) over the specific production factors that influence the distribution of outcomes. Consequently, these goals can be chosen to ensure both a reduction in expected damages and an expected improvement in water quality, and to ensure that the outcome is economically superior to all other possible outcomes. Obviously, complete control over the distribution of outcomes is possible only when goals are specified for each input and technology choice that influences runoff. However, adequate control (in terms of the criteria described above) is possible if goals are chosen for those producer choices that are most correlated with runoff, and if any pollution-increasing substitution effects (i.e., when producers switch to alternative technologies or inputs that may generate more pollution) are limited or of little consequence. Moreover, these goals are advantageous because they can be set deterministically, making it easier to verify whether or not the goals are met. In contrast, it may take years to obtain a large enough sample to determine if a mean ambient water quality goal is achieved (Horan, 1998).

For simplicity, we focus on three types of goals in this report: mean ambient-based goals, mean runoff-based goals, and input- and technology-based goals.[6] The cost-effective outcomes based on mean ambient goals are denoted CE(a), cost-effective outcomes based on expected runoff goals are denoted CE(r), and cost-effective outcomes based on input use and technology are denoted CE(x). All cost-effective plans are characterized by conditions similar to the efficiency conditions discussed earlier, with the expected social benefits of pollution reduction measured in terms of the policy goals.[7] Thus, nonpoint policies must encourage three types of responses for least-cost control: (1) reduction (increase) in the use of variable inputs that increase (mitigate) runoff, (2) adoption of appropriate technologies, and (3) appropriate land-use decisions at the extensive margin (decisions about whether or not to bring land into or out of production, and what to do with land that is taken out of production—for example, plant trees, grass, etc.). The mathematical conditions describing the cost-effective solutions are provided in Appendix 2B.

because designing policies to control aspects of the probability distributions of ambient water quality or runoff is not the same as designing policies to control expected damages. The probability distributions associated with damages, ambient pollution, and runoff are not the same, and control of one distribution does not necessarily imply control of the other.

[6] Many other types of physical goals exist. Even though expected runoff goals are not preferred relative to other goals, we include these goals in our discussion because runoff reduction is often an important goal in practice (e.g., EPA-USDA, 1998).

[7] In addition, analogous definitions for marginal, inframarginal, and extramarginal acreage exist for cost-effective solutions, where social costs of pollution are defined in terms of policy goals.

### Second-Best Policies and Outcomes

Cost-effective and efficient outcomes provide benchmarks from which to gauge the economic performance of alternative policies. That is, these outcomes define actions that would optimally be taken to satisfy NPS pollution policy goals in an ideal world where the set of policy instruments is not restricted and when there are no transactions costs (e.g., costs associated with implementing, administering, and enforcing policies, as well as the costs of obtaining information to design policies) associated with implementing optimally designed policies. Obviously, transactions costs and policy limitations are important and should not be ignored when designing policies. In practice with these limitations and costs, the best possible outcome would achieve policy goals at the lowest cost, given the types of instruments that are used and the costs associated with using them. Such an outcome is generally referred to as **second-best**.[8] While second-best policies are optimal in practice, their economic performance in the sense of being able to achieve a goal at least cost is still measured relative to the ideal of a cost-effective or efficient baseline. This provides a useful method of comparison between alternative types of policies, especially when data on transactions costs are unavailable (as they often are).

## Characteristics of Nonpoint-Source Pollution Influence Policy Design

The characteristics of nonpoint-source pollution (unobservable runoff, natural (weather-related) variability, site-specific nature, etc.) influence how various policy options for controlling NPS pollution might

[8] We use the term “second-best” somewhat loosely. Technically, efficient policies are first-best. Cost-effective policies are also second-best because they are not efficient. For simplicity and consistency, we make a distinction between cost-effective and alternative second-best policies.

**Table 2-1—Example policy objectives**

| Objective | Definition |
|---|---|
| Efficiency | Maximize aggregate farm profits less expected damages |
| Cost-effectiveness: | |
| Mean ambient target | At least cost, reduce expected ambient pollution levels to a specified level. Such an outcome is denoted CE(a). |
| Mean runoff targets | At least cost, reduce expected runoff to specified levels. Such an outcome is denoted CE(r). |
| Input and technology targets | At least cost, achieve input use and technology adoption goals. Such an outcome is denoted CE(x). |
| Second-best | When restricted to policy instruments that are not capable of achieving policy goals at least cost, instruments can be set at levels that achieve the goals at the lowest cost possible for those instruments. The set of policy instruments may be restricted to reduce administrative and enforcement costs, and to be informationally less intensive and applied more uniformly across producers. |

perform. The impacts of these characteristics on policy performance will be dealt with more fully in the following chapters, but it is useful to provide an introductory discussion here.

### *Observability of runoff and loadings*

Agricultural nonpoint-source pollution is difficult to measure or to observe. Most problematic from a policy standpoint is the inability of policymakers (as well as farmers) to observe runoff from a field and loadings into water systems. In addition, monitoring the movement of nonpoint-source pollutants is impractical or prohibitively expensive. Impacts on ambient water quality can be observed. However, because NPS pollution is generated over the land surface and enters water systems over a broad front, and because of the natural variability of the pollution process, these measures of ambient quality do not indicate where pollutants enter the water resource or from which cropland they originate.

The inability to observe loadings would be mitigated if there were a strong correlation between ambient quality and some observable aspect of production. For example, the quality of a shallow aquifer that is entirely overlain by cropland is directly related to how the fields are managed. A policy could then be directed at the production process with a reasonable expectation of the water quality impacts. However, such correlations are extremely unlikely, and where relationships can be established, they are unlikely to hold up across a range of conditions. Because a regulator cannot infer producers' actions by observing the state of water quality, the policymaker is uncertain as to whether poor water quality is due to the failure to take appropriate actions or to undesirable states of nature, like excessive rainfall (Malik, Larson, and Ribaudo, 1994).

Finally, production inputs critical for forecasting NPS pollution may also be unobservable or prohibitively expensive to monitor. For example, there is a close correlation between chemical contamination of groundwater and the amount of a chemical applied and soil type. Chemical characteristics of the pesticide, soil characteristics, and depth to groundwater can all be easily determined. However, application rates and timing are generally not observable to a regulating agency without costly and intrusive monitoring. Producers have a special knowledge about their operations that they may not be willing to share with potential regulators.

### *Natural variability and pollution flows*

Nonpoint-source pollution is influenced by natural variability due to weather-related events (e.g., wind, rainfall, and temperature). As a result, a particular policy will produce a distribution of water quality outcomes (Braden and Segerson, 1993). This by itself does not preclude *ex ante* efficiency through the use of standard policies. However, it greatly complicates policy design. For example, nearly all soil erosion occurs during extremely heavy rain events. Practices that control erosion from "average" rainfalls but fail under heavy rain events will likely be ineffective in protecting water resources from sediment. In addition, natural variability may limit the effectiveness of models in predicting water quality from production decisions since runoff and loadings are not observable.

The natural variability of the NPS pollution process limits policies from being able to achieve ambient or runoff targets at least cost. By nature, policies produce a distribution of results. Therefore, policy must specify both the runoff or ambient targets and the frequency at which those goals are achieved (Shortle 1987, 1990). For example, a nitrogen control policy may require that an ambient goal of 10 mg/liter be met for 75 percent of the samples taken over the course of a year.

### *Heterogeneous geographic impacts*

The characteristics of nonpoint-source pollution vary by location due to the great variety of farming practices, land forms, climate, and hydrologic characteristics found across even relatively small areas. This site-specific nature of NPS pollution has important policy implications. For example, even if models could be developed to measure runoff and loadings, they would have to be calibrated for the site-specific qualities of each individual field. The information required for such calibration would be significant, and possibly unavailable. Therefore, spatial characteristics of cropland and transport/dispersion of pollutants introduce additional uncertainties into the estimation of loadings into water resources (Miltz, Braden, and Johnson, 1988). Policy tools flexible enough to provide cost-effective pollution control under a variety of conditions would outperform tools that are not self-adjusting (Braden and Segerson, 1993).

### *Transboundary effects*

The effects of agricultural nonpoint-source pollution can often be felt far from their source. Chemicals with long half-lives and sediment (pollutants that tend to maintain their properties in a water environment) can affect water users far from where they originate. For example, much of the atrazine and nitrates that enter the Gulf of Mexico each year via the Mississippi River are applied to cropland in the Upper Corn Belt States of Minnesota, Iowa, and Illinois (Goolsby and other, 1993).

### *Uncertain water quality damages*

As with most types of pollution, the economic damages associated with water quality impairment are often difficult to observe or to ascertain. Knowledge of the relationship between economic damages and water pollution is essential for establishing water quality goals or incentive levels that maximize societal welfare. The impacts of pollution on water quality are often nonmarket in nature. For example, nitrates in the Chesapeake Bay are believed to reduce submerged aquatic vegetation (SAV) levels. There is no market for SAV; however, SAV has economic value because it provides habitat for economically valuable fish populations, among other things. Without organized markets, information on the value of water quality may be difficult to obtain. Even if these impacts are observed and can be attributed to specific sources, valuation requires the use of a nonmarket valuation technique such as travel cost or contingent valuation (Ribaudo and Hellerstein, 1992). Such exercises are both time consuming and costly, and the reliability of such techniques is in question. Therefore, a cost-effective policy that achieves a more easily measured physical goal might be more practical than one based on estimated damages.

### *Time lags*

The movement of a pollutant off a field to the point in a water system where it imposes costs on water users may take a considerable amount of time. Time lags of this sort have two policy implications. First, observed ambient water quality conditions may be the result of past management practices, or of polluters no longer in operation. Second, the results of a policy may not be immediately apparent, making it difficult to assess its effectiveness.

## Selecting Policy Tools for Reducing Nonpoint-Source Pollution

Policy instruments at the Federal, State, or local level for controlling water pollution fall into five general classes: (1) economic incentives, (2) regulation, (3) education, (4) liability, and (5) research and development.[9] Policymakers must consider a number of important economic, distributional, environmental, and political characteristics when selecting an instrument.

### *Economic performance*

The instruments differ in their ability to maximize net social benefits by correcting an externality. Some may be able to achieve only a second-best solution because external pollution costs are not fully accounted for when production decisions are made. The policy instruments also distribute costs of pollution control differently between polluters and the rest of society. For example, subsidies place the burden of pollution control on taxpayers, while taxes place the burden on polluters.

[9] These instruments will be covered in detail in the following chapters.

The **basis** of a policy instrument is the point in the pollution stream to which the instrument is applied, and has a bearing on the performance of the instrument. Instruments can be applied either to farmers' actions or to the results of their actions. For point sources, the preferred basis is discharge because it is directly related to water quality and because it is easy to observe (Baumol and Oates, 1988). However, the choice is not so clear for nonpoint sources. Proposed bases include ambient pollution levels, expected runoff levels, input use, technology, and output. Bases most closely correlated to water quality (runoff and ambient pollution) are preferred to those that are more indirectly related, such as agricultural output (Braden and Segerson, 1993). Directing policy instruments at bases that are only indirectly correlated with water quality may lead to unrelated effects and inefficient management.

### Administration and enforcement costs

The costs of administering a water quality protection policy and enforcing it are related to a variety of factors, including the nature of the pollution problem, the legal system, and the information required to implement an instrument efficiently. These costs have particular importance for policies aimed at controlling nonpoint-source pollution. Nonpoint runoff is difficult to monitor due to its stochastic and diffuse nature. Likewise, measurements of ambient concentration and chemical loss may be subject to error. In addition, while it is straightforward to monitor the use of purchased inputs, not so the use of all polluting inputs such as manure applications. If the cost of detecting noncompliance is too high, polluters will be able to skirt the policy (Braden and Segerson, 1993). Administration and enforcement costs need to be weighed against the potential environmental benefits of the policy.

### Flexibility

Policy instruments may be flexible both for producers and for the managing agency. A policy instrument is flexible for producers if they are able to reduce pollution control costs by adjusting their production and pollution control decisions in the face of changing economic conditions (such as changes in input and output prices or the availability or new technologies), changing environmental conditions (such as rainfall), and site-specific physical conditions (such as slope and soil quality) and, in at least some situations, still meet policy goals.

A policy instrument is flexible for a resource management agency if it continues to provide the proper signal or incentive to producers in the face of changing economic and environmental relationships that underlie its construction. An inflexible instrument would require an adjustment to continue meeting a policy goal if conditions changed. Adjusting a policy instrument may be costly. The resource management agency is left with a choice of either efficiency loss if the instrument is not adjusted, or potentially high transactions costs if the rate is adjusted. Flexibility is an empirical issue that has not been addressed in the nonpoint literature and that will likely depend on specific circumstances. However, in the face of changing economic and environmental relationships, flexibility may be increased if the agency has fewer instruments to adjust. For example, if a single runoff tax can be used to provide the same results as two input taxes, then the runoff tax would be more flexible for the agency because fewer adjustments would be required if relationships changed. In this report, we focus primarily on flexibility with respect to producers, and also on flexibility for an agency in terms of how many instruments are required.

### Innovation

A policy instrument should encourage and reward farmers for using their unique knowledge of the resource base to meet policy goals (Shortle and Abler, 1994; Bohm and Russell, 1985; Braden and Segerson, 1993). Instruments that provide these incentives are more likely to achieve cost-effective control than those that do not.

### Political and legal feasibility

Even though several policy instruments are equally capable of an economically efficient outcome, they may not be perceived as equal for legal or political reasons. The difficulty in observing nonpoint runoff may be a source of legal problems for instruments using runoff or ambient quality as a base. For example, it may be difficult to hold individual farmers legally responsible for observed water quality damages when the sources of NPS pollution cannot be observed. The stochastic nature of nonpoint pollution also makes it difficult to accurately infer damages or runoff based on farm practices (Shortle, 1984; Tomasi, Segerson, and Braden, 1994). In addition, ambient pollution levels may be the result of past management decisions due to time lags in pollution transport. Thus, absent contributors to the ambient pollution level may cause current farming operations to be unfairly punished.

An instrument's political feasibility may be related to ethical and philosophical arguments. For example, in the absence of any water quality laws, the right to clean rivers and streams is not assigned to any group. As a result, an activity such as farming, which produces runoff that can pollute rivers and streams, is not obliged to consider the impacts of its activities. This can be considered an implicit "right" to pollute. Taxes and permits may then be politically unpopular among farmers because these instruments implicitly shift pollution "rights" from farmers to the users of water resources. Alternatively, a subsidy to reduce pollution implicitly affirms the producer's "right" to pollute. Those who seek cleaner water must pay for it. This position may be protested by the victims of pollution, who believe they have a right to clean water, and by other industries that are legally required to reduce pollution. In an era of widespread anti-tax sentiment, a tax-based environmental policy may be impossible to implement, despite efficiency considerations.

## Choosing an Appropriate Institutional Structure

Are water quality programs best implemented at the local, State, regional, or national levels, and with what type of coordination across levels? Major government activities for any environmental policy include setting standards, selecting appropriate policy tools, implementation, and enforcement. Policy can be centralized, where all activities are handled by the Federal Government (possibly with local input), or localized with State or local governments having all or most of the responsibilities. Braden and Matsueda (1997) present a set of environmental problem characteristics that can be used to determine the level of governance best able to provide efficient control. Some of these characteristics involve the nature of the pollution problem, while others concern the abilities of different levels of government to operate programs efficiently.

### *Regional differences*

Nonpoint-source pollution varies by place due to the great variety of farming practices, land forms, climate, and hydrologic characteristics found across even small areas. Since benefits and costs of policies are likely to vary along with these factors, a policy should take them into account (Shortle, 1995; Sunding, 1996)

Both centralized and local governments can tailor policy to local conditions. A centralized policy would require that decisionmakers obtain much local information, resulting in potentially high transaction costs (Fort, 1991). These costs could be reduced by using more uniform standards and policies, but at the expense of greater inefficiency. Local governments are better able to set standards that reflect local demands for water quality and to take into account local economic and physical characteristics in setting policy (assuming they have the resources to acquire information). Many Federal environmental laws recognize local variation and pass some of the responsibilities for standard setting and policy implementation to the States.

### *Influence of interest groups*

Through bargaining and other influences, special interest groups can often influence both the quantity and price of public goods such as environmental quality, resulting in an inefficient allocation of resources (Braden and Matsueda, 1997). Decentralized policies give local special interests more influence because they are proportionately larger and face less competition in the local economy than in, say, the national economy (Fort, 1991; Esty, 1996; Braden and Matsueda, 1997; Lester, 1994). For instance, threats to move an important local industry could influence local leaders to underprovide environmental protection. Agricultural interests often have an important voice in agricultural areas where nonpoint-source pollution would be generated. Centralized policies would have an advantage over local policies in counteracting local special interests.

### *Uncertainty*

Uncertainty can take two forms: uncertainty about the causes and consequences of an environmental problem, and uncertainty about the consequences of public policies (Braden and Matsueda, 1997). Nonpoint-source pollution is characterized by uncertainty in production, movement, and impacts on water quality. Uncertainty about the causes and consequences of an environmental problem can be reduced through research. Environmental research generates information that is a pure public good, and so its cost is appropriately spread across the entire population (Braden and Matsueda, 1997). Centralized government is seen as better suited to provide this research because the research results would be available to all (Braden and Matsueda, 1997; Esty, 1996).

An advantage posited for decentralized policies is that States will try different policy approaches, and through these "experiments" the most effective policies will emerge (Esty, 1996; Braden and Matsueda, 1997); however, no single researcher controls experiments, tabulates results, or draws conclusions. A centralized system, with State cooperation, could be in a better position to conduct policy experiments. Centralized leadership could set the limits or scope of policy experiments; spread the costs of research, data gathering, and interpretation; and subsidize sharing of information with States (Braden and Matsueda, 1997).

### *Transboundary issues*

Local jurisdictions have an advantage over more centralized authorities in developing policies tailored to local conditions. However, some agricultural pollutants travel long distances (Goolsby and others, 1993). Under these circumstances, the smaller the jurisdictions providing environmental protection, the greater the chance for the costs of pollution and benefits of policies to fall outside (Esty, 1996). Locally based policies tend to account only for local benefits and costs (Braden and Matsueda, 1997; U.S. Congress, 1997). The result is an inefficient allocation of resources. A basic principle of federalism is that economic efficiency in the provision of public goods is best served by delegating responsibility for the provision of the good to the lowest level of government that encompasses most of the associated benefits and costs (Shortle, 1995). With pollutants that travel long distances, this prinicple could enlist very large regions.

Widespread, routine transboundary problems require policies that apply widely; a centralized authority might achieve economies of scale in setting standards and implementing policy (Braden and Matsueda, 1997). Efficient policy requires that all beneficiaries be considered, even if they reside outside a government's area of control (Esty, 1996; Fort, 1991). This includes those who suffer from pollutants transported outside the area of jurisdiction (transboundary supply) as well as consumers who reside outside the jurisdiction area but value water quality within (transboundary demand). Following the rule of fiscal federalism, defining jurisdiction on the basis of all beneficiaries leads to more centralized policies.

Of course, local jurisdictions could handle transboundary problems through interstate agreements and compacts. However, very seldom have States come together without Federal prodding to address regional water quality issues, despite common goals and the fact that an individual State may be unable to meet water quality goals without better control of interstate pollution. For example, the Northeastern States for years tried to get Midwestern States to better control sulfur emissions that were causing acidification of Northeastern lakes, without success (Price, 1982). Only four compacts or inter-regional commissions are devoted to water quality and the management of major rivers that cross State boundaries: Delaware River Commission (New York, New Jersey, Delaware, and Pennsylvania), Interstate Sanitation Commission (New Jersey, New York, and Connecticut), Ohio River Valley Water Sanitation Commission (Pennsylvania, Ohio, Illinois, Indiana, Kentucky, New York, Virginia, and West Virginia), and the Susquehanna River Basin Commission (New York, Pennsylvania, and Maryland) (EPA, 1995).

Watersheds cross political jurisdictions, and by all accounts, are the most appropriate geographic units for implementing specific water quality protection plans (EPA-USDA, 1998). A watershed is the geographic area in which water, sediments, and dissolved materials drain to a common outlet—a point on a larger stream, a lake, an underlying aquifer, an estuary, or an ocean. A watershed is also known as a drainage basin. If watersheds are defined as the USGS 8-digit hydrologic unit, of which there are 2,111 in the contiguous 48 States, 35 percent of the land area in these States is within watersheds that span more than 1 State (table 2-2). Thus, States must cooperate to manage land (for nonpoint-source pollution) and point sources in order to provide efficient water quality protection in many watersheds. Without cooperation, a single State may be unable to address a water quality problem in such watersheds, or may have to implement unnecessarily stringent measures to achieve water quality goals.

If larger rivers are of most concern and the targets for water quality improvement, then individual State actions will require widespread cooperation. For example, defining watersheds at the 6-digit hydrologic unit code (600 watersheds), 71 percent of the land area is within watersheds that cross State boundaries. At

**Table 2-2—Land area in watersheds that cross State borders, 48 contiguous States**

| State | 8-digit[1] | 6-digit | 4-digit |
|---|---|---|---|
| | | *Percent* | |
| Alabama | 42 | 85 | 100 |
| Arizona | 21 | 53 | 60 |
| Arkansas | 44 | 71 | 100 |
| California | 17 | 40 | 53 |
| Colorado` | 43 | 92 | 92 |
| Connecticut | 90 | 100 | 100 |
| Delaware | 68 | 100 | 100 |
| Florida | 20 | 36 | 36 |
| Georgia | 32 | 66 | 100 |
| Idaho | 40 | 100 | 100 |
| Illinois | 43 | 58 | 68 |
| Indiana | 51 | 66 | 100 |
| Iowa | 39 | 100 | 100 |
| Kansas | 41 | 100 | 100 |
| Kentucky | 48 | 91 | 100 |
| Louisiana | 34 | 44 | 56 |
| Maine | 13 | 16 | 16 |
| Maryland | 87 | 100 | 100 |
| Massachusetts | 74 | 100 | 100 |
| Michigan | 17 | 42 | 66 |
| Minnesota | 22 | 60 | 63 |
| Mississippi | 37 | 100 | 100 |
| Missouri | 52 | 92 | 100 |
| Montana | 22 | 60 | 60 |
| Nebraska | 47 | 65 | 71 |
| Nevada | 31 | 85 | 100 |
| New Hampshire | 81 | 100 | 100 |
| New Jersey | 60 | 75 | 100 |
| New Mexico | 48 | 78 | 100 |
| New York | 41 | 67 | 70 |
| North Carolina | 51 | 58 | 58 |
| North Dakota | 27 | 61 | 87 |
| Ohio | 39 | 65 | 65 |
| Oklahoma | 50 | 91 | 100 |
| Oregon | 29 | 84 | 88 |
| Pennsylvania | 35 | 85 | 100 |
| Rhode Island | 100 | 100 | 100 |
| South Carolina | 48 | 74 | 100 |
| South Dakota | 35 | 87 | 100 |
| Tennessee | 59 | 100 | 100 |
| Texas | 21 | 39 | 41 |
| Utah | 42 | 95 | 100 |
| Vermont | 82 | 94 | 94 |
| Virginia | 57 | 100 | 100 |
| Washington | 19 | 29 | 38 |
| West Virginia | 62 | 100 | 100 |
| Wisconsin | 41 | 89 | 100 |
| Wyoming | 36 | 96 | 100 |
| U.S. | 35 | 71 | 79 |

[1] Hydrologic Unit Codes used to identify watersheds in the United States. There are 2,111 8-digit watersheds, 600 6-digit watersheds, and 99 4-digit watersheds.

the 4-digit hydrologic unit level (99 watersheds), 79 percent of land area is in watersheds that cross State boundaries. However, there are few examples of such cooperation. The transaction costs of all the agreements necessary to handle all transboundary problems would be enormous. Centralization, depending on the skill of the policymakers and implementers, may reduce transaction costs and maintain efficiency.

Water quality spillovers can occur not only over space, but also over time. Some dimensions of environmental quality are not easily restored once degraded. Ground water is an example. Concern for future generations is more difficult to incorporate in local policy decisions than at the national level (Oates and Schwab, 1988). An individual's children and their offspring will probably live elsewhere, creating a myopic view of environmental quality that could lead to a suboptimal provision of environmental goods (Oates and Schwab, 1988). Centralized decision-making, in principle, is better suited to internalize the "demand" for current environmental standards from future generations.

### *Economies of scale*

A single product standard set at the national level imparts efficiency benefits on manufacturers who then do not have to meet 50 different State standards. For nonpoint-source pollution, economies exist in the technical expertise to set, monitor, and enforce standards (Braden and Matsueda, 1997; Fort, 1991; Esty, 1996; Smith and other, 1993). For example, there is no need to conduct research on the health and environmental effects of a pesticide in all 50 States. These effects would be the same everywhere. Data collection, testing quality assurance/quality control, fate and transport studies, epidemiological and ecological analyses, and risk assessments all represent highly technical activities in which expertise is important and scale economies are significant (Esty, 1996). In fact, the Federal Government provides much information in the areas of water quality monitoring, land use surveys, health studies, and fate and transport studies that States can use to implement their own programs. States may implement their own research programs to get finer detail, but many States lack the technical capacity to do this (Lester, 1994; Esty, 1996).

### *Interjurisdictional competition*

One of the classic arguments for centralized control is the so-called "race to the bottom" (Esty, 1996; Oates and Schwab, 1988), which holds that State and local governments engage in active competitions with each other for new business (jobs and tax base). Under these conditions, local officials do not propose tax rates or environmental regulations that go much beyond those in "competing" States. The costs of environmental regulation on business are more easily translated into monetary terms than benefits, and are concentrated on a relatively few entities (polluters) (Esty, 1996). The end result is that all States underprovide environmental quality, leading to an inefficient allocation of resources.

This scenario is hotly debated (Revesz, 1992; Esty, 1996). While the argument is plausible, there has been little systematic analysis of whether States actually engage in distortionary competition (Oates and Schwab, 1988; Braden and Matsueda, 1997). Oates and Schwab used a neoclassical model to demonstrate that a race to the bottom is not inevitable and that States can provide the optimal level of public goods. However, this result requires some stringent assumptions, including no distortionary taxes (such as taxes on capital) and a "benevolent" bureaucracy that seeks and knows public values for nonmarket goods. When these assumptions do not hold, a race to the bottom cannot be discounted (Oates and Schwab, 1988).

Lester (1994) identified a number of States that are capable of providing a higher level of environmental quality but have chosen not to do so. The differences between States in the level of environmental protection is not, however, evidence of destructive competition. In fact, different environmental protection across States could be consistent with an efficient market solution. Destructive competition would be evident if environmental protection, given citizen demand for quality and the costs of control, is less than optimal in at least some States. There is currently no empirical data that destructive competition occurs (U.S. Congress, 1997).

## Summary and Conclusions

Nonpoint-source pollution is produced at inefficiently high levels because farmers do not generally account for pollution's costs to others when making their production decisions. An ideal goal of policy to control such pollution is to get farmers to consider these external costs (an efficient solution). However, such a goal may be unattainable if the governing agency has limited information about damages or the pollution process. As an alternative, policy may be designed to attain specific water quality or input and technology goals (e.g., a limit on mean ambient pollution levels, or a limit on mean runoff levels) at least cost (a cost-effective solution).

The characteristics of nonpoint-source pollution pose particular challenges for designing and implementing efficient or cost-effective pollution control policies. These characteristics include:

- runoff and loadings cannot be observed;
- NPS pollution is characterized by natural (weather-related) variability;
- characteristics of NPS pollution vary over geographic space;
- pollution can travel long distances;
- water quality damages are difficult to observe or to measure;
- considerable time lags complicate assessment.

Policymakers have a number of policy tools available for addressing agricultural nonpoint-source pollution, including economic incentives (taxes, subsidies, permit trading), standards, liability, education, and research. Which ones are most appropriate depends on a number of economic, distributional, and political considerations, including how well the tool achieves the policy goals, the costs of administering and enforcing the policy, the ability of the policy to adjust to different economic and physical conditions, how well the policy encourages innovation, and political and legal feasibility.

Another issue is whether a policy is best implemented by local, State, or national levels of government, and the degree of coordination necessary for effective pollution control. The success of a particular institutional structure is influenced by geographic variability in nonpoint-source pollution characteristics, the ability of special interest groups to move a policy away from the optimal economic solution, the ability of the institutional structure to address uncertainty, the geographic scale of the pollution problem, and the likelihood of interjurisdictional competition's resulting in less-than-optimal policies.

# Appendix 2A—Nonpoint-Source Pollution Policy Conditions for Efficiency[10]

Consider a situation in which a particular resource (e.g., a lake) is damaged by a single residual (e.g., nitrogen) from nonpoint sources of pollution. The ambient concentration of the pollutant is given by

$$a = a(r_1, r_2, ..., r_n, W)$$

where $a$ is the ambient concentration ($\partial a/\partial r_1 > 0$), $r_i$ ($i$ = 1, 2,..., $n$) is runoff from the $i$th agricultural production site, and $W$ reflects the influences of weather and other stochastic events on the transport process.

Runoff from a particular site is a function of the production activities on that site. Production activities will involve either choices made along a continuum (e.g., chemical application rates, irrigation rates, etc.) or discontinuous choices (e.g., tillage, chemical application methods, crop choice, rotation, etc.). Continuous choices are assumed to correspond to variable input use. Denote the ($m$ x 1) vector of inputs chosen for use on the $i$th acre by $x_i$. For simplicity, discontinuous choices are represented by a scalar, $A_i$, which is referred to as the technology in use. For example, $A_i = 1$ might correspond to the production of continuous corn with no-till tillage, $A_i = 2$ might correspond to production using a rotation of corn and soybeans, using mulch-till tillage, etc. Runoff from the $i$th site is given by the runoff function, $r_i = r_i(x_i, A_i, v_i)$, where $v_i$ is a site-specific random variable describing natural occurrences affecting runoff. No assumptions are made about the relation between the technology's productivity and runoff (i.e., a more productive technology does not necessarily correspond to greater or lesser runoff levels). Instead, a variety of possibilities could arise, depending on the technology.

Farmers are assumed to be risk-neutral. The expected profit from site $i$ for any choice of inputs and technology is given by the strictly concave function $\pi_i(x_i, A_i)$. Larger values of $i$ are assumed to correspond to sites with less productive land and that are more conducive to NPS pollution generation (e.g., soil type, slope of land, distance from water etc.).[11] For simplicity, farmers in this particular region are assumed not to have any collective influence on the prices of inputs or outputs, and that input and output markets are free from distortions. Finally, the economic cost of damages caused by pollution is given by $D(a)$ ($D'$, $D'' \geq 0$)

An *ex ante* efficient allocation maximizes the expected net surplus (quasi-rents, less environmental damage costs) to society (Just, Hueth, and Schmitz 1982, Freeman 1993). The appropriate objective function, restricted on technology, is the following[12]

$$J(A) = \underset{x_{ij},n}{Max} \sum_{i=1}^{n} p_1(x_i, A_i) - E\{D(a)\}$$

The necessary conditions for a maximum are

$$\frac{\partial J}{\partial x_{ij}} = \frac{\partial p_i}{\partial x_{ij}} - E\{D'(a)\frac{\partial a}{\partial r_i}\frac{\partial r_i}{\partial x_{ij}}\} = 0 \quad \forall i, j \qquad \text{(2A-1)}$$

$$\frac{\Delta J}{\Delta n} \approx p_n(x_n, A_n) - E\{\Delta D(a)\} \approx 0 \qquad \text{(2A-2)}$$

where $\Delta D(a) = D(a(r_1, \ldots, r_n, W)) - D(a(r_1, \ldots, r_{n-1}, W))$ (i.e., the difference in damages with and without site $n$).

Condition (2A-1) equates marginal net private benefits from the use of $x_{ij}$ with expected marginal external damages from the use of the input. If the externality is ignored, then condition (2A-1) is violated and the level of input use for inputs that increase runoff (i.e., inputs for which $\partial r_i / \partial x_{ij} > 0$) will be too high while the level of input use for inputs that decrease runoff (i.e., inputs for which $\partial r_i / \partial x_{ij} < 0$) will be too low. The resulting runoff levels will be too high and a Pareto maximum will not be achieved.

Condition (2A-2) describes the incremental impact of the $n$th site on expected net benefits. If the nth site is defined optimally, then the addition of any other site

[10] This appendix develops the mathematical foundations for Pareto efficiency. The basic framework closely follows that of Horan et al. (1998a) and Shortle et al. (1998a).

[11] In reality, the relationship between site productivity and conduciveness to runoff is not one-to-one. A more realistic specification would include separate, jointly distributed indices for these two attributes. See Shortle and others (1997) for a model with dual indices and Horan and others (1998a) for a formal derivation that makes use of information on both productivity and conduciveness to runoff.

[12] Following most NPS pollution literature, it is assumed that society is risk-neutral. A more general model would choose input levels in an expected utility framework.

will have a negative incremental impact. Positive profits are earned on the marginal site, $n$, and all infra-marginal sites because $E\{\Delta D(a)\} > 0$. If external damages are ignored, the amount of land able to produce profitably is greater than otherwise. The result is increased runoff due to increased production in the industry, and hence ambient pollution levels will be higher than is economically efficient. Together, conditions (2A-1) and (2A-2) define the efficient scale of production for the marginal site.

Finally, the optimal technology vector, $A^*$, is determined by solving for an efficient allocation for each possible value of A and comparing expected net benefits. Technology $A^*$ is more efficient than technology $A'$ when $J(A^*)$ - $J(A') > J(A')$. Thus, the optimal technology vector satisfies the condition

$$J(A^*) - J(A') \geq 0 \;\forall A' \qquad \text{(2A-3)}$$

which reduces to

$$\begin{aligned} &\pi(x_i(A_i),A_i^*) - \pi_i(x_i(A_i'),A_i') \geq E\{D(a(r_1^*,\ldots,r_i^*,\ldots,r_n^*,W))\} \\ &-E\{D(a(r_i^*,\ldots,r_{i-1}^*,r_i(x_i(A_i'),A_i',v_i),r_{i+1}^*,\ldots r_n^*,W))\} \quad \forall A_i' \end{aligned} \qquad \text{(2A-4)}$$

where $r_i^* = r_i(x_i(A_i^*), A_i^*, \nu_i)$. The choice of technology will be inefficient if the externality is ignored, due to the technology's impacts on runoff.

## Appendix 2B— Cost-Effective Policy Design

Pollution control policies can be designed to minimize costs (or, equivalently, to maximize net benefits in the absence of damages) subject to a constraint based on the ambient pollution level, a set of constraints based on runoff when the damage and pollutant transport relationships are unknown, or input and technology constraints. Policy designed in this situation will generally not be efficient; however, it can lead to a cost-effective solution as input use is allocated among farms at least cost to meet an exogenously specified constraint. Specifically, the resource management agency's problem can be written as

$$\max_{x_{ij},A_i,n} \quad J = \sum_{i=1}^{n} \pi_i(x_i, A_i) \qquad \text{(2B-1)}$$

subject to a constraint or set of constraints based on an ambient or runoff target(s).

The degree of reliability with which water quality or runoff targets are to be achieved must be specified because a particular policy will produce a distribution of outcomes (Braden and Segerson 1993). Many constraints have been proposed in the literature, however, two are of particular interest (Beavis and Walker 1983; Beavis and Dobbs 1987; Shortle 1990; Horan 1998):[13]

$$E\{a\} \leq a_o \qquad \text{(2B-2)}$$

$$E\{r_i\} \leq r_{io} \;\; \forall i \qquad \text{(2B-3)}$$

where $a_0$ is an exogenously chosen ambient target, and $r_{i0}$ is an exogenously chosen runoff target for the $i$th site in production.

### *A Cost-Effective Solution Based on a Mean Ambient Target*

The Lagrangian corresponding to the maximization of (2B-1) subject to (2B-2) is

$$L = \sum_{i=1}^{n} \pi_i(x_i, A_i) + \lambda\,[a_0 - E\{a\}]$$

where λ is the Lagrangian multiplier. Assuming an interior solution, the first-order conditions with respect to input use and the number of sites are

$$\frac{\partial L}{\partial x_{ij}} = \frac{\partial \pi_i}{\partial x_{ij}} - \lambda E\{\frac{\partial a}{\partial r_i}\frac{\partial r_i}{\partial x_{ij}}\} = 0 \quad \forall i, j \qquad \text{(2B-4)}$$

$$\frac{\Delta L}{\Delta n} \approx \pi_n - \lambda E\{\Delta a\} \approx 0 \qquad \text{(2B-5)}$$

where $\Delta a = a(r_1, \ldots, r_n, W) - a(r_1, \ldots, r_{n-1}, W)$. These conditions have the same interpretation as conditions (2A-1) and (2A-2), except that marginal costs are defined in terms of the constraint as opposed to damages. The shadow value λ is the value of the optimal tax/subsidy rate when farmers and the resource management agency share the same expectations about the nonpoint process.

[13] Constraints may also be of the form $P(a \leq a_0) = 1 - \alpha$ where α is the probability that $a$ will exceed the target (Beavis and Walker 1983). We do not focus on this type of constraint because it would be difficult to use in practice (Shortle 1990).

Finally, the optimal technology vector, $A^*$, is determined by solving for an optimal allocation for each possible value of $A$ and comparing aggregate profits. The optimal technology vector satisfies the condition

$$L(A^*) - L(A') \geq 0 \forall A' \tag{2B-6}$$

In particular, the following condition must hold

$$\begin{aligned} &\pi_i(x_i(A_i^*), A_i^*) - \pi_i(x_i(A_i'), A_i') \geq \\ &\lambda E\{a(r_1^*, \ldots, r_i^*, \ldots, r_n^*, W)\} \\ &- \lambda E\{a(r_1^*, \ldots, r_{i-1}^*, r_i', r_{i+1}^*, \ldots, r_n^*, W)\} \quad \forall i, \forall A_i' \end{aligned} \tag{2B-7}$$

Conditions (2B-6) and (2B-7) have the same interpretation as (2A-3) and (2A-4).

The cost-effective solution will generally not be efficient (Horan, 1999).[14] Moreover, use of a smaller ambient target, $a_1 < a_0$, may not result in a more efficient outcome if the variability of $a$ is increased as a result of using the smaller target (Shortle, 1990).

### *A Cost-Effective Solution Based on Mean Runoff Targets*

The Lagrangian corresponding to the maximization of (2B-1) subject to (2B-3) is

$$L = \sum_{i=1}^{n} \pi_i(x_i, A_i) + \sum_{i=1}^{n} \lambda[r_{i0} - E\{r_i\}]$$

where $\lambda_i$ is the Lagrangian multiplier for the $i$th runoff constraint. Assuming an interior solution, the first-order conditions with respect to input use and the number of sites are

$$\frac{\partial L}{\partial x_{ij}} = \frac{\partial \pi_i}{\partial x_{ij}} - \lambda_i E\{\frac{\partial r_i}{\partial x_{ij}}\} = 0 \quad \forall i, j \tag{2B-8}$$

$$\frac{\Delta L}{\Delta n} \approx \pi_n - \lambda_n[r_{n0} - E\{r_n\}] \approx 0 \tag{2B-9}$$

[14] It is not possible to attain an efficient solution unless (1) there is only one site with one production choice, or (2) the covariance between marginal damages and marginal ambient pollution is zero for all sites and inputs.

Condition (2B-8) has the same interpretation as condition (2A-1), except that marginal costs are defined in terms of the constraint as opposed to damages. The shadow values $\lambda_i$ equal the optimal tax/subsidy rates when farmers and the resource management agency share the same expectations about the nonpoint process.

Assuming the constraint (2B-3) is satisfied as an equality, condition (2B-9) reduces to a zero profit condition for the marginal site. However, since input use in the cost-effective solution will generally differ from input use in the competitive solution, the marginal site in the cost-effective solution will generally differ from the marginal site in the competitive solution.

Finally, the optimal technology vector, $A^*$, is determined by solving for an optimal allocation for each possible value of $A$ and comparing aggregate profits. The optimal technology vector satisfies the condition

$$L(A^*) - L(A') \geq 0 \; \forall A' \tag{2B-10}$$

In particular, the following condition must hold

$$\begin{aligned} &\pi_i(x_i(A_i^*), A_i^*) - \pi_i(x_i(A_i'), A_i') \geq \\ &\lambda_i E\{r_i^*\} - \lambda_i E\{r_i' \quad \forall i, \forall A_i' \end{aligned} \tag{2B-11}$$

Conditions (2B-10) and (2B-11) have the same interpretation as (2A-3) and (2A-4).

The cost-effective solution will generally not be efficient.[15] Moreover, use of smaller runoff targets, $r_{i1} < r_{i0}$ $\forall i$, may not result in a more efficient outcome if the variability of $r_i$ for some $i$ is increased as a result of using the smaller targets (Shortle 1990).

### *A Cost-Effective Solution Based on Input Use*

Input goals may be defined in terms of either site-specific input use or aggregate input use within a region such as a watershed. For simplicity, only the former case is considered here. Goals may also be defined either for all inputs that contribute to runoff, or for only a subset of these inputs. For example, nitrogen runoff from agriculture depends not only on the amount of nitrogen applied, but also on plant uptake which is a function of crop yield. Each input

[15] It is not possible to attain an efficient solution unless: (i) there is only one production choice, or (ii) the covariance between marginal damages from runoff and marginal runoff is zero for all sites and inputs.

that influences crop yield will therefore generally influence runoff; however, policy goals may be specified only as reductions in nitrogen fertilizer use.

Let $z_i$ denote the $(m' \times 1)$ vector of inputs for which goals are defined, and let $y_i$ denote the $([m - m'] \times 1)$ vector of inputs for which there is no goal ($x_i' = [y_i, z_i]$). Input-based goals are then defined by

$$z_{ij} \le \bar{z}_{ij} \quad \forall i, j \tag{2B-12}$$

where $\bar{z}_{ij}$ is the target for use of the $j$th input on the $i$th site. The goals defined by (2B-12) are flexible in that they may be site-specific, or they may be uniform across firms within a region (in which case $\bar{z}_{ij} = \bar{z}_{1j}$ $\forall i$, $l$). Moreover, (2B-12) is equivalent to an input reduction goal (for those inputs that increase runoff) or an input expansion goal (for those inputs that reduce runoff), specified in either absolute terms ($z_{ij}^C - z_{ij} \le \bar{A} = z_{ij}^C - \bar{z}_{ij}$ $\forall i,j$, where $z_{ij}^C$ is firm $i$'s competitive level of use of input $j$) or percentage terms ($[z_{ij}^C - z_{ij}]/z_{ij}^C \le P = [z_{ij}^C - z_{ij}]/z_{ij}^C$ $\forall i,j$). An example of the latter goal would be a 25-percent reduction in nitrogen application rates within a region. In this case, the goal is uniform while the input use target $\bar{z}_{ij}$ is, in general, site-specific.

The Lagrangian corresponding to the maximization of (2B-1) subject to (2B-12) is

$$L = \sum_{i=1}^{n} \pi_i(z_i, y_i, A_i) + \sum_{i=1}^{n} \sum_{j=1}^{m'} \lambda_{ij}[z_{i0} - z_i]$$

where $\lambda_{ij}$ is the Lagrangian multiplier for the $j$th input constraint for the $i$th site. Assuming an interior solution, the first-order conditions with respect to input use and the number of sites are

$$\frac{\partial L}{\partial z_{ij}} = \frac{\partial \pi_i}{\partial z_{ij}} - \lambda_{ij} = 0 \quad \forall i, j \tag{2B-13}$$

$$\frac{\partial L}{\partial y_{ij}} = \frac{\partial \pi_i}{\partial y_{ij}} = 0 \quad \forall i, j \tag{2B-14}$$

$$\frac{\Delta L}{\Delta n} \approx \pi_n - \lambda_n[r_{n0} - E\{r_n\}] \approx 0 \tag{2B-15}$$

Condition (2B-13) has the same interpretation as condition (2A-1), except that marginal costs are defined in terms of the constraint as opposed to damages. The shadow values $\lambda_{ij}$ equal the optimal incentive rates for input use.

Assuming the constraint (2B-12) is satisfied as an equality, condition (2B-15) reduces to a zero profit condition for the marginal acre. However, since input use in the cost-effective solution will generally differ from input use in the competitive solution, the marginal site in the cost-effective solution will generally differ from the marginal site in the competitive solution.

Finally, the optimal technology vector, $A^*$, unless it is specified by policy goals, is determined by solving for an optimal allocation for each possible value of $A$ and comparing aggregate profits. The optimal technology vector satisfies the condition (2B-10). In particular, the following condition must hold

$$\pi_i(x_i(A_i^*), A_i^*) - \pi_i(x_i(A_i'), A_i') \ge \sum_{j=1}^{m'} [\lambda_{ij}(A_i^*) x_{ij}(A_i^*) - \lambda_{ij}(A_i') x_{ij}(A_i')] \quad \forall i, \forall A_i' \tag{2B-16}$$

Condition (2B-16) has the same interpretation as (2A-4).

# Chapter 3

# Economic Incentives

*Economic incentive-based instruments, such as taxes or subsidies, are used by policymakers to create prices for the externalities (i.e., economic damages) that farming produces. These policy instruments effectively alter prices in existing markets or create new markets so that producers have incentives to control pollution at socially desirable levels. In this chapter, we detail a variety of economic incentive-based instruments that may be used for nonpoint pollution control and evaluate these instruments according to several criteria related to instrument design, implementation, and the incentives created.*

## Introduction and Overview

Agricultural nonpoint-source pollution occurs at greater levels than are socially optimal because markets fail to accurately relay the social costs of pollution to producers. Economic incentive-based instruments, such as taxes or subsidies, are used by policymakers to create prices for the externalities (i.e., economic damages) that are produced. These policy instruments effectively alter prices in existing markets or create new markets so that producers have incentives to control pollution at socially desirable levels.

Economists have suggested a variety of incentive-based instruments to control nonpoint-source pollution. However, no general comparison of instruments exists. In this chapter, we provide a detailed discussion of a variety of economic incentive-based instruments that may be used for nonpoint pollution control. Specifically, we show that:

- Incentives must be designed to transmit the goals of policymakers. Producers respond differently to various incentives, depending on the *base* to which the incentive is applied (e.g., the incentive base of a fertilizer tax is fertilizer) and the complexity of the instrument.
- Design-based incentives are generally superior to performance-based incentives.
- Second-best, input- and technology-based incentives are most conducive to policy.
- Coordination of existing programs and improved targeting of incentives are needed for further improvements to water quality.
- Properly designed market-based systems may be effective alternatives to existing programs to control nonpoint pollution.

This chapter begins with a general overview of incentives. Next, we review the two main classes of incentive bases: (1) performance-based incentives (i.e., incentives based on runoff, measured ambient concentrations, or damages), and (2) design-based incentives (i.e., incentives based on inputs and technology). (Table 3-1 lists the economic incentives that are covered in this chapter and provides examples of actual application of each.) Within each class, we consider a variety of specific incentive bases and how each has been applied at the Federal level, evaluating each instrument according to (1) the incentives it provides, (2) its relative complexity, (3) informational requirements of a resource management agency in designing the instrument and of producers in using the instrument to evaluate their decisions, (4) flexibility of the instrument to changing economic and environmental conditions, and (5) potential administration and enforcement costs. In addition, we discuss how policy design issues relate to policies that have been implemented at the Federal level (noting that major State policies are similar). Finally, we review two alternative types of incentives—compliance mechanisms and market mechanisms—and discuss practical experience with these pollution control methods.[1]

[1] We limit our focus to nonpoint policies. However, point sources of pollution will influence damages as well. Point source and nonpoint-source pollution control policies should therefore be conjunctive (see Shortle and Abler, 1997; Shortle and others, 1998a).

**Table 3-1—Types of incentives and examples from Federal programs**

| Incentives | Federal program applications |
|---|---|
| Performance-based: | |
| Runoff | None in existence |
| Ambient | None in existence |
| Design-based: | |
| Expected runoff | None in existence |
| Variable inputs | None in existence |
| Technology (i.e., fixed inputs, production techniques, etc.) | USDA Conservation Compliance, Swampbuster, ACP, WQIP, EQIP |
| Acreage at the extensive margin | Conservation Reserve Program |

## Characteristics of Economic Incentives

Policymakers can use economic incentives to create prices for nonpoint pollution externalities so that producers will control pollution at more socially desirable levels. Incentives may alter prices in existing markets (e.g., a nitrogen tax increases the price of nitrogen) or they can create new markets that did not previously exist (e.g., a market for expected runoff levels is created by either taxing expected runoff levels and forcing producers to "buy" expected runoff from society, or by issuing permits for expected runoff levels to producers and allowing them to sell permits among themselves). Profit-maximizing producers are then forced to consider the social cost of pollution when making management decisions. Management choices are then more consistent with society's environmental objectives.

Economic incentives are generally classified as either a tax or a subsidy.[2] In the case of nonpoint pollution, taxes make it more expensive for producers to pollute by increasing the cost of pollution-causing activities. Alternatively, subsidies make it less expensive for producers to not pollute by decreasing the cost of pollution-mitigating activities. The effect of each can be the same, depending on how they are applied.

The major benefit of economic incentive-based policies is that producers can choose whatever strategy is most profitable for them. In addition, producers' strategies can change as relative prices for inputs and outputs change, or as new technologies become available. Pollution abatement costs will generally be lower with incentives than with command and control policies because producers may be able to utilize site-specific attributes (which a resource management agency may have limited information about) to their advantage in reducing control costs. In addition, innovators may have an incentive to develop and market new approaches that help producers reduce pollution control costs.

### *Two Types of Taxes and Subsidies*

For simplicity, we focus on constant, per-unit incentives (e.g., a sales tax) and lump sum incentives. For a tax, total payments equal the (constant) *per-unit tax* rate multiplied by the tax base. The relationship between total subsidy receipts and the subsidy base is slightly different. A subsidy can be used to provide the same outcome as a tax with the same per-unit rate. However, subsidy payments are often determined relative to a benchmark level. For example, a subsidy applied to fertilizer use might be based on a reduction in use from a specific level. The greater the reduction in total fertilizer use, the greater the subsidy. No subsidy would be provided if there were no reduction in fertilizer use.

A *lump sum* instrument is a fixed tax or subsidy that can be used to influence discrete choices or to determine the distributional outcomes of policies. With respect to discrete choices, lump sum instruments can be made contingent on particular actions. For example, a producer can be paid a lump sum amount if he/she adopts a particular tillage practice, and paid nothing if adoption does not occur. Alternatively, lump sum instruments that are not contingent on particular actions are not applied to a base and therefore do not influence marginal incentives.

### *Subsidies Versus Taxes for Pollution Control*

Taxes and subsidies can be designed to have the same effect on producers' production and pollution control decisions. However, taxes and subsidies will have different impacts on farm profits and on a resource management agency's budget. Taxes will generally reduce farm profits and increase agency budgets, while subsidies will have the opposite effect. However, it is possible to use taxes without reducing farm profits by pro-

[2] The permit price in a market for pollution permits essentially operates as a tax.

***A subsidy implicitly supports the view that polluters are not responsible for pollution. Instead, polluters are given the "right" to pollute and society must pay polluters for cleaner water. An alternative view is that society holds the "rights" to cleaner water and that polluters should pay for pollution control (i.e., the "polluter pays" principle). This alternative view is supported by taxes and regulatory policies, and has shaped many point-source programs. For example, point-source control policies under the Clean Water Act hold polluters responsible for treatment costs.***

viding producers with a lump sum refund of expected tax payments.

Subsidies require specification of a benchmark level from which they will be determined (e.g., with a point-source emissions subsidy, firms receive a larger subsidy the further emissions are reduced below the benchmark). The specification of this benchmark may create perverse incentives. For example, suppose abatement of point-source pollution is to be subsidized. Establishing a firm-specific pollution abatement benchmark at current discharge levels would penalize firms that have already undertaken pollution abatement. For example, a firm that has been able to reduce emissions to 4 tons on its own would have a 4-ton benchmark while a firm that has not reduced emissions and produces 8 tons would have an 8-ton benchmark. For a given pollution level, the firm with the 8-ton benchmark will receive a larger subsidy and be rewarded for not attempting to reduce pollution on its own. Therefore, establishing a benchmark at current discharge levels would create an immediate, perverse incentive for a firm to produce as large a discharge as possible in order to elevate its benchmark (Baumol and Oates, 1988). Finally, when subsidies are used, society (as opposed to the polluter) must pay for pollution control.

## Performance-Based Incentives

Performance-based incentives are taxes or subsidies pursuant to a firm's production and pollution control decisions. Two outcomes of producers' decisions are the most logical targets of incentives for reducing nonpoint water pollution: runoff from a field and ambient water quality conditions.

In theory, a tax or subsidy can be based on how much runoff leaves a site so that the external cost of pollution is considered by producers when they make their production and pollution control decisions.[3] This is akin to an effluent tax on factory discharge. Unfortunately, runoff cannot be monitored at reasonable cost given current monitoring technologies. Only with advances in monitoring technologies will runoff-based instruments become viable policy tools for controlling nonpoint pollution.

Even if runoff was observable, its suitability as an incentive basis would be limited by the natural variability of runoff and other nonpoint processes. Optimally, incentives provide producers with information about the impacts of their choices on expected damages from pollution, and assign them responsibility accordingly. However, a single runoff-based incentive rate can only provide information about how individual choices are expected to impact runoff and not damages. This is because a runoff-based incentive induces producers to consider the impacts of their choices on mean runoff levels, and choices made to achieve a particular mean runoff level do not correspond to a unique level of expected damages. Instead, these choices could have a variety of unintended impacts to damages, due to random events. Thus, runoff-based incentives will not generally provide producers with enough information to accurately consider the external costs of each of their decisions. Similar results occur when trying to achieve an ambient water quality goal at least cost.

Ambient-based incentives are based on the ambient pollution levels in the water resources affected by farming's activities. These incentives are (seemingly) advantageous for two reasons. First, economic theory suggests instrument bases (ambient pollution levels) should be close to the externality (damages from pollution). Second, ambient pollution can be monitored without the resource management agency having to observe the actions of each producer. However, these advantages quickly disappear when informational requirements and other complexities associated with policy design are taken into account (table 3.2).

[3] Incentives can also be applied to each farm's pollution loading to the stream, which runs off from fields influenced by transport characteristics. The results are similar. The only difference is that a loadings incentive requires the producer to determine some transport impacts, while the runoff incentive places this burden on the regulator.

**Table 3-2—Evaluation of performance-based incentives**

| Criteria | Runoff-based | Ambient-based | | |
|---|---|---|---|---|
| | | Efficient, CE(r), CE(x) | Cost-effective: CE(a) | Second-best (uniform, limited information) |
| Incentives provided | N/A | Instrument exists only under very restrictive assumptions. | Poor<br><br>Not efficient. Exists only when producers are all risk-neutral and when producers and the resource management agency share identical expectations. Additional instruments required for optimal entry/exit. | Poor<br><br>Not cost-effective. Additional instruments required for optimal entry/exit. |
| Overall complexity | N/A | N/A | Medium-High<br><br>A producer must be able to evaluate how he/she and others influence the incentive base. | Medium-High<br><br>A producer must be able to evaluate how he/she and others influence the incentive base. |
| Information required by producers | N/A | N/A | High<br><br>Each producer needs information about production and runoff characteristics of all producers and pollution transport. | High<br><br>Each producer needs information about production and runoff characteristics of all producers, and pollution transport. |
| Flexibility | N/A | N/A | High<br><br>Producers can respond to changing market conditions. Agency has to set one rate for each producer. | High<br><br>Producers can respond to changing market conditions. Agency only has to set one uniform rate. |
| Administration/ enforcement costs | Currently prohibitive | N/A | High<br><br>High information costs. Potentially high monitoring costs in some cases. | Medium-High<br><br>Medium to high information costs. Potentially high monitoring costs. |

N/A = not applicable because instrument is impractical. These rankings are subjective, based only on theoretical properties as opposed to empirical evidence. A more reliable table would be based on empirical results that compare each type of policy according to a consistent modeling framework that is representative of the nonpoint problem.

### Incentives Provided by the Instruments

Ambient-based incentives can be designed to achieve an efficient or cost-effective (CE) outcome only under highly restrictive conditions (Horan and others, 1998a,b). For example, a CE outcome designed to achieve a mean ambient pollution target—a CE(a) outcome—can be achieved only when producers are risk-neutral and producers and the resource management agency have the same expectations about the nonpoint process. The ambient tax/subsidy rate that leads to the CE(a) outcome is uniformly applied across producers and equals the social cost of a marginal increase in mean ambient pollution levels. Such a tax/subsidy rate transmits the policy goal of the policymakers to the producers. A CE(a) outcome is possible in this case because the goals of producers would coincide with those of policymakers (i.e., to control mean ambient pollution levels at least cost). When the expectations of producers and the resource management agency differ, a cost-effective solution cannot be achieved because the goals of the producers will differ among themselves and from the goals of the resource management agency (see appendix 3A).

Risk-averse producers will not like the additional risk (due to the natural, weather-related uncertainty associated with ambient pollution levels) that ambient-based incentives create. Instead, risk-averse producers will prefer design-based instruments that can produce the same social outcome and have the same expected impacts on profitability. Moreover, ambient-based instruments cannot produce the CE(a) outcome when used alone. This is because producers' production and pollution control choices have uncertain impacts on ambient pollution levels, creating risk that cannot be adequately controlled with an ambient-based incentive alone (Horan and others, 1998b).

When some producers are risk-averse and/or when ambient-based incentives cannot be designed to accurately transmit the resource management agency's goals, then ambient-based incentives can only be second-best (i.e., achieve policy goals at least cost given risk aversion and heterogeneous expectations about the nonpoint process). Potentially high transaction costs may necessitate that second-best incentives be applied at uniform rates across producers.

Ambient pollution levels depend on the mix of sites in production in the region. If a suboptimal mix of sites is in production, then each producer will face the wrong incentives for input use and technology choices (since these incentives depend on ambient pollution levels, which depend on the mix of sites in production), and equilibrium ambient pollution levels will be suboptimal relative to CE(a) or second-best levels.

By themselves, ambient-based incentives do not provide incentives for optimal entry and exit.[4] Additional lump sum instruments, however, will induce optimal entry and exit into production in the region (Horan and others, 1998a). The lump sum incentives would take the form of a tax applied to producers who produce on extramarginal sites (if they do not produce on this land, they pay no tax) or a subsidy given producers who voluntarily retire extramarginal acreage.[5] It is not necessary to apply lump sum taxes or subsidies to producers who produce on marginal and inframarginal land unless their decision to produce is influenced by the magnitude of the tax. A lump sum refund of their expected tax would reduce their expected tax burden to zero without compromising cost effectiveness.

### Relative Complexity of the Instruments

Ambient-based instruments are complex from a producer's perspective because producers must be able to evaluate how their actions and the actions of others affect the incentive base (since the incentive base depends on group performance). Given the large number of nonpoint polluters that may exist within a region, such instruments are likely to be too complex for producers to make accurate evaluations. In that case, producers will receive incorrect incentives from ambient-based instruments.

### Informational Requirements

Ambient-based instruments place a large informational burden on producers. To attain a CE(a) or second-best outcome, each producer would have to have information about the actions of other producers and how these actions affect ambient pollution levels or expected damages. Given the large number of nonpoint polluters that may exist within a region, producers are not likely to acquire such information.

[4] *Entry and exit* refer to the process of production sites being entered into or removed from production. Optimal entry and exit occurs when production occurs at positive levels on the marginal and inframarginal sites, but ceases on extramarginal sites.

[5] For example, the Conservation Reserve Program (CRP) uses subsidies to induce producers to retire environmentally sensitive land from production.

The resource management agency also has significant informational requirements. For any performance-based or design-based instrument, the agency must have information about producers and the pollution process so that it can evaluate the impact of the policy instrument on ambient water quality (more information is better, although policies can be designed with less than perfect information). With ambient-based instruments, the agency has the additional burden of having to understand how producers evaluate the impacts of their decisions on water quality. In other words, the agency must understand each producer's belief structure about the nonpoint process. This added requirement is likely to limit the ability of the resource management agency to construct CE(a) or second-best ambient-based incentives.

#### *Flexibility Provided by the Instrument*

Producers have flexibility in their production and pollution control decisions under ambient-based incentives in that they may utilize any private knowledge they may have to further reduce costs, or they may alter their decisions as economic and environmental conditions change. The resource management agency may have more flexibility than with some design-based instruments because there is only a single incentive rate to adjust as underlying economic and environmental relationships change. In contrast, several types of rates must be altered as underlying relationships change when incentives are applied to several inputs.

#### *Administration and Enforcement Costs*

Information costs associated with setting ambient-based instruments at appropriate levels may be significant. Monitoring costs depend on how easy it is to monitor ambient pollution levels or damages. Monitoring may be relatively easy in some cases (small reservoir or lake) but relatively difficult in others (ground water or major river with many tributaries).

### Application of Performance-Based Incentives

Performance-based incentives have not generally been applied in the United States. One possible exception is a tax being used in Florida to reduce phosphorus discharges to the Everglades. The Everglades Forever Act calls for a uniform, per-acre tax on all cropland in the Everglades Agricultural Area. The tax was implemented in 1994 at a rate of $24.89 per acre per year, and will increase every 4 years to a maximum of $35.00 per acre by 2006 unless phosphorus is reduced 25 percent basinwide (State of Florida, 1995). Reductions in phosphorus are determined through monitoring of runoff water that collects in drainage ditches. This type of tax is based on acres of cropland—a design base; however, its application depends on phosphorus levels—a performance base. The tax creates the incentive to adopt best-management practices, and also for producers to apply pressure on recalcitrant neighbors. The number of producers is not so large that free-riding is much of a problem.

This tool is flexible in that producers are not restricted in how they manage their operations to meet the phosphorus goal. However, the basis upon which the tax is placed—acres of cropland—is not necessarily consistent with the goal of the tax, phosphorus reduction. A more efficient approach (and potentially practical, given the small number of polluters) would be to tax phosphorus loads directly.

## Design-Based Incentives

Design-based incentives are based on a producer's variable input use and production technology.[6] Producers have no uncertainty about design-based incentives when making decisions, and each producer's decisions may be observed by a resource management agency (although not always easily). However, input use and technology are further removed from damages than with performance-based instruments. Design-based incentives can be based on expected runoff (which is estimated based on inputs and technology) or on inputs and technology directly. After evaluating each subclass, we discuss practical applications of design-based incentives.

### Expected Runoff-Based Incentives

Expected runoff levels from cropland may be estimated (before runoff actually occurs) with a simulation model that incorporates all production and pollution control decisions. The incentive base (expected runoff) is therefore design-based because it depends explicitly on inputs and technology (table 3-3).[7]

[6] The inputs and technology targeted by policy may include aspects of pollution control that are unrelated to production.

[7] There may be legal problems with basing permits on the resource management agency's expectations about runoff instead of actual runoff, especially given the limited ability of modelers to accurately predict runoff from input use.

**Table 3-3—Evaluation of expected runoff-based incentives**

| Criteria | Efficient, CE(a), or CE(x) | Cost-effective: CE(r) | Second-best (imperfect information, uniform) |
|---|---|---|---|
| Incentives provided | Instrument exists only under very restrictive assumptions. | Good<br><br>Cost-effective but not efficient. Additional instruments required to ensure cost-effective entry/exit. | Fair<br><br>Cost-effective but not efficient. Additional instruments targeted at entry/exit may increase efficiency. |
| Overall complexity | N/A | Medium-High<br><br>Instrument may be site-specific or uniform. Producers must evaluate how their production and pollution control decisions influence the instrument base. | Medium-High<br><br>Instrument may be site-specific or uniform. Producers must evaluate how their production and pollution control decisions influence the instrument base. |
| Information required by producers | N/A | Medium<br><br>Producers need information about their own runoff process. However, this information can be provided by the resource management agency | Medium<br><br>Producers need information about their own runoff process. However, this information can be provided by the resource management agency. |
| Flexibility | N/A | High<br><br>Producers are able to respond to changing market conditions. Agency has to set only one rate for each producer | High<br><br>Producers are able to respond to changing market conditions. Agency has to set only one rate for each producer. |
| Administration and enforcement costs | N/A | High<br><br>A simulation model must be developed to determine expected runoff levels for each acre in production. All input and technology decisions must be monitored. | Medium-High<br><br>Use of limited information may reduce costs. A simulation model must be developed to determine expected runoff levels for each acre in production. All input and technology decisions must be monitored. |

N/A = not applicable because instrument not practical. These rankings are subjective, based only on theoretical properties as opposed to empirical evidence. A more reliable table would be based on empirical results that compare each type of policy according to a consistent modeling framework that is representative of the nonpoint problem.

### *Incentives Provided by the Instrument*

Expected runoff-based tax/subsidy rates can be designed to achieve an efficient outcome, (i.e., to control expected damages), CE(a) outcome (i.e., to control expected ambient pollution levels), or CE(x) outcome (i.e., to control input use and technology) only under highly restrictive conditions (Shortle and others, 1998b).[8] This is because expected runoff-based instruments provide producers with incentives to control mean runoff levels from their field, and these incentives generally differ from the goal of policymakers who wish to achieve an efficient, CE(a), or CE(x) outcome (Shortle, 1990; Horan 1998). Expected runoff-based instruments can be designed to achieve a CE(r) outcome (to control runoff at least cost) because the goals of producers then coincide with those of policymakers. The optimal incentive rate in this case would be site-specific, equal to the social value of a marginal increase in mean runoff from the site.

An expected runoff-based instrument will be effective only if producers understand how their production and pollution control decisions influence expected runoff. This information may be provided to producers by the resource management agency in the form of a tax or subsidy schedule based on input and technology choices, or the agency may provide producers with access to the runoff simulation models. Differing expectations about the runoff process are not important here as they were with ambient-based instruments because the incentive is based on the resource management agency's expectations. There would be no benefit to producers from using their own expectations.

Political or legal reasons or transaction costs may prevent a resource management agency from implementing site-specific incentives. Instead, a single incentive rate may be applied uniformly to each site. No matter what policy goals are chosen, a uniform instrument provides incentives for producers to reduce expected runoff levels at least cost. Therefore, the instrument is a cost-effective method of achieving a set of mean expected runoff levels, even if the mean levels achieved do not correspond to the policy goals (i.e., a uniform incentive always leads to a CE(r) outcome). A cost-effective uniform incentive rate equals the average of the expected marginal social costs created by runoff from each site, plus (in the case that policy goals are not to control expected runoff) an additional term (a risk premium) to account for the risk associated with controlling expected runoff as opposed to the policy goal (Shortle and others, 1998b). A uniform expected runoff incentive is not likely to reduce administration costs significantly because the resource management agency would have to construct a model of each site to determine compliance and all inputs and technologies would have to be monitored for use in the model.

If expected runoff incentive rates are set at levels to attain the CE(r) outcome, then the mix of production sites may not be cost-effective because of suboptimal entry and exit (see appendix 3A). The cost-effective mix of sites may be obtained by providing lump sum incentives to producers who produce on marginal or extramarginal sites. It is not necessary to provide lump sum subsidies to producers on inframarginal sites unless, in the case of expected runoff taxes, their decision to produce is influenced by the magnitude of the tax. However, a lump sum refund of these producers' taxes would reduce their tax burden to zero without compromising efficiency.

Second-best policies may be designed when producers retain private information. The resource management agency may have imperfect information about production practices, land productivity, and other site-specific characteristics that affect runoff or economic returns. Producers may be reluctant to truthfully provide any private information to the resource management agency for fear that this information might be used against them in the design of environmental policy. While it may be possible to develop a cost-effective incentive scheme that induces producers to truthfully report their private information, it is implausible due to large informational requirements and related monitoring and enforcement costs (see Shortle and Abler (1994) for the case of such an input-based incentive scheme).

Alternatively, it is possible to design incentives to attain a second-best benchmark that allows producers to retain their private information.[9] In the absence of administration and enforcement costs, policy designed with limit-

[8] Specifically, an efficient rate exists when either (1) the producer makes only a single decision that influences runoff or (2) the covariance between marginal damages and marginal runoff levels is zero for each input (Shortle and others, 1998b).

[9] Policies designed under imperfect information cannot be designed to attain a specific outcome. With limited information, the resource management agency can design policy based only on how it expects producers to react. Therefore, policy would have to be designed to attain an expected outcome.

ed site-specific information will generally be less efficient than policy designed under perfect information. However, given the large costs of obtaining site-specific information, policy designed where producers retain their private information may actually be optimal.

The efficiency of a second-best incentive can be increased if additional instruments are used for entry and exit. Lump sum incentives for achieving optimal entry and exit would take the form of a tax applied to producers producing on extramarginal sites (if they do not produce on this land, they pay no tax) or a subsidy applied to producers who voluntarily retire extramarginal sites (e.g., USDA's Conservation Reserve Program). It is not necessary to provide lump sum taxes or subsidies to producers who produce on marginal and inframarginal sites unless, in the case of an expected runoff tax, their decision to produce is influenced by the magnitude of the tax. However, a lump sum refund of their expected tax would reduce their expected tax burden to zero without compromising optimality.

### *Overall Complexity of the Instrument*

An expected runoff incentive is administratively complex because input use and technology must be monitored for each site in order to determine expected runoff levels (using a simulation model). In addition, the resource management agency would have to develop a model to simulate runoff from each agricultural production site.

### *Informational Requirements*

Each producer must understand how production and pollution control decisions affect runoff if the instrument is to be effective. Information on the relationship between runoff and production and pollution control decisions may be provided to each producer by the resource management agency. To attain a cost-effective outcome, the resource management agency requires perfect information about production and runoff characteristics. Less information is required in designing policies to achieve second-best outcomes. However, efficiency is increased as more information is used to design policy.

### *Flexibility Provided by Instrument*

An expected runoff-based incentive is fairly flexible. Producers have flexibility in that they may utilize any private knowledge they may have to further reduce costs, or they may alter production decisions as economic and environmental conditions change. The resource management agency may have more flexibility than with some design-based instruments because there is only a single instrument base (expected runoff levels) for which incentive rates must be altered as underlying economic and environmental relationships change. When incentives are applied to several inputs, several types of rates must be altered as underlying relationships change.

### *Administration and Enforcement Costs*

Monitoring costs are high for expected runoff-based instruments because the use of each input and technology must be monitored to determine (through the use of a simulation model) expected runoff. Also, providing producers with information about runoff relationships for each production site (by providing access to simulation models) would likely be expensive. Information and administration costs would be higher with site-specific instruments than with uniform or second-best instruments designed using less-than-perfect information.

## Input- and Technology-Based Incentives

The second subclass of design-based incentives is based more directly on inputs and technology (Shortle and Abler, 1994). A summary of input- and technology-based instruments (not including expected runoff-based instruments), according to evaluative criteria, is presented in table 3-4.

### *Incentives Provided by the Instruments*

Input- and technology-based incentives can be designed to achieve an efficient or any type of cost-effective outcome (i.e., a CE(a), CE(r), or CE(x) outcome; see table 2-1). The reason is that input and technology choices, while not always equivalent to specific policy goals, are the means by which a resource management agency can achieve its goals. For example, if a resource management agency had absolute control over agricultural production in a region and wanted to achieve an efficient outcome, it would do so by specifying input use and technologies for the region.

Instruments must target all inputs and technology choices to attain an efficient, CE(a), or CE(r) outcome. The cost-effective incentive rate would be site-specific,

**Table 3-4—An evaluation of input- and technology-based incentives**

| Evaluative criteria | Efficient or cost-effective: CE(a), CE(r), or CE(x) | Second-best (i.e., uniform, limited set of inputs, imperfect information) |
|---|---|---|
| Incentives provided | Good | Fair |
| | Additional instruments are needed to ensure optimal entry/exit. | Not efficient. Additional instruments required for optimal entry/exit. |
| Relative complexity | Medium | Low |
| | Efficiently or cost-effectively designed instrument is site-specific and applied to each input and technology choice. Producers can easily evaluate instruments. | Incentives applied only to a few input and technology choices, and may be uniformly applied to all producers. Producers can easily evaluate instruments. |
| Information required by producers | Low | Low |
| | Producers need information about only their own production processes. | Producers need information about only their own production processes. |
| Flexibility | Medium | Medium-High |
| | Producers are able to respond to changing market conditions. Incentives for each production and pollution control decision. Resource management agency must set multiple rates for each producer. | Producers are able to respond to changing market conditions. Incentives for only some production and pollution control decisions. Resource management agency must set multiple rates for each producer. |
| Administration and enforcement costs | Medium-High | Low-Medium |
| | Site-specific incentive applied to each production and pollution control choice requires an extensive amount of monitoring. | Costs are reduced the more uniformly the incentives are administered, the fewer inputs are targeted, and the less site-specific information the resource management agency pursues. |

Note: These rankings are subjective, based only on theoretical properties as opposed to empirical evidence. A more reliable table would be based on empirical results that compare each type of policy according to a consistent modeling framework that is representative of the nonpoint problem.

equaling the expected social cost of a marginal increase in the use of the input (Shortle and others, 1998a; Shortle and Abler, 1997). Note that the social cost of a marginal increase in the use of an input is negative for those inputs that decrease pollution (e.g., a nitrogen inhibitor). The use of such inputs should be subsidized.

The use of per-unit, input-based incentives alone will not create the incentives necessary to induce producers to adopt the efficient technology (e.g., placing the appropriate taxes on variable inputs may not induce a switch from conventional tillage to conservation tillage).[10] If a suboptimal technology is used, then input use may also be suboptimal since all production decisions are interdependent. Therefore, the optimality of input taxes/subsidies is conditional on the technology chosen. Additional instruments, targeted at technology, are required to attain the efficient, CE(a), or CE(r) outcome.

Lump sum incentives that are contingent on technology choices can produce optimal adoption. For example, a lump sum tax can be applied to producers who adopt a suboptimal technology, or a lump sum subsidy can be applied to producers who adopt the optimal technology. If there are adjustment costs to technology adoption, a cost-sharing approach can also be used to induce adoption.

[10] This is because the choice of production technology has a nonmarginal impact on damages, but the linear instruments only account for the marginal impacts of each producer's choices.

Producers may have available to them a variety of crop production and pollution control technologies and will likely be operating with a suboptimal technology prior to the implementation of nonpoint pollution control policies. The cost of switching to an alternative technology may be significant. Nowak (1987) identifies 15 constraints to adoption (see box, p. 50), most having to do with the costs of obtaining information, management and capital constraints, and perceptions about risk. These constraints explain the frequent use of suboptimal crop management strategies.

Additional instruments may be necessary to ensure optimal entry and exit. The use of input and lump sum technology taxes/subsidies may not result in efficient or cost-effective entry and exit into the region. It may therefore be necessary to apply a lump sum tax/subsidy to producers producing on extramarginal sites to ensure optimal entry and exit. Otherwise, there will be an inefficient mix of sites in production, resulting in too much pollution for the region. The optimal lump sum tax would be applied to producers who produce on extramarginal sites and would ensure that they do not earn after-tax profits on these sites. Alternatively, a lump sum subsidy could be given to producers who retire extramarginal sites. The optimal value would ensure that these producers are better off when they do not produce on extramarginal sites. Lump sum subsidies to producers on marginal and inframarginal sites are unnecessary unless, in the case of input and technology taxes, their decision to produce is influenced by the magnitude of the other taxes. However, a lump sum refund of these producers' taxes would reduce their tax burden to zero without further compromising efficiency.

The resource management agency may have imperfect information about production practices, land productivity, and other site-specific characteristics that affect runoff or economic returns, and producers may be reluctant to truthfully reveal any private information. The agency may therefore have to design a second-best benchmark that allows producers to retain their private information.[11] In the absence of administration and enforcement costs, policy designed with limited site-specific information will generally be less efficient than policy designed under perfect information. However, given the large costs of obtaining site-specific information, policy designed when producers retain their private information may actually be optimal.

[11] See footnote 10.

Political or legal reasons or costs may limit the ability of a resource management agency to implement site-specific incentives for each input that contributes to pollution. Instead, incentives may be applied uniformly across sites and applied to only a few inputs, reducing administration costs. The choice of inputs to target could be based on ease of observation or measurement. Some management practices, such as the rate at which chemicals are applied, are very difficult to observe without intensive and obtrusive monitoring.

An optimal uniform incentive rate equals the average of the expected marginal social costs created by the input use at each site, plus adjustments to account for the average marginal impacts of input substitution on expected social costs and profit levels (Shortle and others, 1998a). The adjustments are needed because placing incentives on the most easily observed inputs can lead to substitution distortions and undesirable changes in the input mix (Eiswerth, 1993; Stephenson, Kerns, and Shabman, 1996). For example, a tax on herbicides would reduce herbicide use, but may increase mechanical cultivation and soil erosion, which in turn has undesirable impacts on water quality. The resource management agency would have to carefully consider the management alternatives to the undesirable practices, and have in place economic incentives or other measures to counter any undesirable characteristics of the alternatives.

The efficiency of second-best, input-based incentives can be increased if additional instruments are used for technology adoption and entry/exit. Specifically, lump sum technology taxes/subsidies could be administered to all producers to ensure optimal technology adoption, and lump sum taxes/subsidies could be administered to producers on extramarginal sites to ensure proper entry and exit (e.g., the CRP). The efficiency gain from using these lump sum instruments diminishes as the uniformity of the lump sum taxes/subsidies grows. Lump sum tax refunds could be provided to producers on marginal and inframarginal sites, reducing their tax burden to zero without further compromising efficiency.

### *Relative Complexity of the Instrument*

Input-based instruments are relatively simple because they are applied as an excise tax/subsidy on variable inputs. Technology-based instruments, since they are lump sum, are also relatively simple. However, the site-specific nature of efficient or cost-effective instruments increases their administrative complexity.

Second-best instruments are designed to be more simple. Other things equal, uniform instruments will be administratively less complex than site-specific instruments, and instruments applied to only a few inputs will be less complex to administer than instruments applied to all inputs. Finally, instruments designed with limited information will be less complex from an administrative perspective.

### *Informational Requirements*

The resource management agency must have perfect information about production and runoff functions for any efficient or cost-effective solution that attempts to control nonpoint pollution. However, second-best policies may be designed with only limited information about site-specific characteristics. Producers have no special informational requirements with input- and technology-based incentives.

### *Flexibility Provided by Instrument*

Producers have flexibility in their production and pollution control decisions under input- and technology-based incentives in that they may utilize any private knowledge they may have to further reduce costs, or they may alter their decisions as economic and environmental conditions change. A resource management agency would have less flexibility with these instruments since a number of incentive rates would have to be adjusted as underlying environmental and economic relationships change.

### *Administration and Enforcement Costs*

Administration, monitoring, and enforcement costs are relatively high for all efficient or cost-effective input- and technology-based instruments due to their site-specific nature and the necessity of monitoring each input and technology used. Second-best instruments are less costly to apply because they do not have to be site-specific, nor does every input and technology choice have to be monitored for each producer. Information costs may also be reduced with second-best policies.

## Application of Design-Based Incentives

Studies of actual or proposed economic incentive-based policies for reducing agricultural nonpoint-source pollution are limited. Only a few States have used input-based incentives, and their impact on agricultural nonpoint pollution problems has not been determined. Economists must therefore rely on simulative modeling techniques to gauge how these instruments might perform. Technology subsidies (cost-sharing and incentive payments) and land retirement (extensive margin) subsidies (CRP) are the only tools that have been extensively used for reducing agricultural nonpoint-source pollution.

### *Input-Based Incentives*

The empirical literature on input-based incentives consists primarily of different incentive policy simulations (e.g., Abrahams and Shortle, 1997; Babcock et al., 1997; Helfand and House, 1995; Larson, Helfand, and House, 1996; Tsai and Shortle, 1998; Weinberg and Wilen, 1997). These studies all contend that incentives can be targeted at a limited number of inputs (such as irrigation water or chemical use) and still achieve environmental goals with cost effectiveness. However, the choice of base is important. Cost effectiveness is increased if incentive bases are highly correlated with policy goals (Russell, 1986), and if the incentives encourage producers to reduce sufficiently the use of pollution-causing inputs while not using more of other pollution-causing inputs or less of pollution-mitigating inputs. For example, Helfand and House (1995) and Larson, Helfand, and House (1996) explore alternative tax policies to limit aggregate expected nitrogen runoff levels from lettuce production in the Salinas Valley, California. They find that taxing irrigation water is more cost-effective than taxing nitrogen fertilizer inputs, and almost as cost-effective as regulating both inputs optimally. Water had a higher correlation with runoff, and producers were more likely to use less water than less nitrogen when faced with a given incentive. Peters, McDowell, and House (1997) also found that tax rates on nitrogen fertilizer must be high to reduce expected nitrogen loss due to an inelastic demand for fertilizer.

The uniformity of incentives across sites is also an issue. Helfand and House (1995) determined the use of uniform input taxes within a region to be almost as cost-effective as site-specific taxes. This result is not supported by others, however. Babcock and others (1997), Russell (1986), and Tsai and Shortle (1998) find that targeting incentives to specific sites may significantly outperform uniform approaches due to local geographic and hydrologic conditions. These studies, however, did not consider the additional administrative and information costs associated with improved targeting.

Finally, empirical research suggests that input-based incentives are likely to have only indirect effects on technology choices (or other types of discrete choices such as crop choice and rotation) (Hopkins, Schnitkey, and Tweeten, 1996; Taylor, Adams, and Miller, 1992). If a set of input taxes induces an inefficient set of discrete choices, then input use is likely to remain inefficient as well. For example, inefficient input use can be expected if an input tax policy induces farmers to adopt an inefficient crop rotation. This is because the (efficient) tax rates will fail to provide farmers with appropriate incentives under the production relationships that correspond to an inefficient rotation. The result may be inefficiently high pollution levels (Hopkins, Schnitkey, and Tweeten, 1996; Taylor, Adams, and Miller, 1992).

### *Technology Adoption Subsidies*

USDA and most States have long offered farmers incentive payments for the adoption of conservation practices. Historically, payments were based on the installation cost of primarily structural practices, such as terraces. More recently, the advent of programs such as the Water Quality Incentive Program (WQIP) and Environmental Quality Incentive Program (EQIP) have made payments available for nonstructural management practices, such as conservation tillage. These payments are designed to offset any private losses a farmer may incur by adopting the practice, any increased risk (in terms of uncertain yields) over the first several years of implementation, and any other short-term adoption constraints (see box, "Constraints to Adoption of Alternative Management Practices").

The incentive payments offered by USDA are technology-intensive in that they focus on management practices. Efficiency will be increased if technology-based incentives are used in conjunction with input-based incentives. In order for the short-term subsidy to elicit a change in technology, it must equal the present value of the stream of expected net losses from adopting the practice, if the practice reduces profits. If the practice increases profits, then the subsidy's value is simply that amount necessary to overcome adoption constraints.

Even though incentive payments have been an important tool for many programs, their effectiveness may be limited. USDA financial assistance programs indicate that practice profitability, rather than short-term subsidies, is the most important factor for long-term adoption. The Rural Clean Water Program of the 1980's demonstrated that cost-shared practices had to be attractive on their own merits (EPA, 1990). In a study of soil conservation decisions in Virginia, Norris and Batie (1987) found that farm financial factors, as opposed to cost-sharing, were the most important influences on the use of conservation practices. This suggests that either subsidy levels were not high enough or that subsidies were not offered long enough to be effective.

WQIP incentives may also have been inadequate for encouraging many farmers to adopt practices less damaging to water quality. A 1994 Sustainable Agriculture Coalition study found that WQIP incentive payments were too low in some regions to secure the adoption of recommended practices, including waste management systems, conservation cover, conservation tillage, critical area planting, filter strips, pasture and hayland management, pasture and hayland planting, planned grazing systems, stripcropping, nutrient management, pest management, and recordkeeping (Higgins, 1995).

An Economic Research Service (ERS) study (Cooper and Keim, 1996) used the results of farmer surveys from the Eastern Iowa-Illinois Basin, Albemarle-Pamlico, Georgia-Florida, and Upper Snake Area Study projects (joint ERS-Natural Resources Conservation Service (NRCS)-U.S. Geological Survey projects to study relationships between production practices and water quality) to model the probability of adopting a preferred farming practice as a function of WQIP incentive payments. The practices studied included split fertilizer applications, integrated pest management, legume crediting, manure crediting, and soil moisture testing. Results suggested that adoption rates of 12 to 20 percent could be achieved with no payment, indicating that some practices were profitable on their own merit in some regions. However, the adoption rate would not increase beyond 30 percent with the actual WQIP payments of $10/acre. A substantial payment increase would be required to encourage 50-percent adoption for any of the prac-

## Constraints to Adoption of Alternative Management Practices

Nowak (1991) identified 15 constraints to adoption:

1. Basic information about the practice is lacking. Producers do not have adequate information to assess the economic and agronomic properties of a practice, and how the practice might meet overall goals (e.g., profitability or stewardship). A producer will not blindly adopt a new practice without adequate information.

2. Cost of obtaining information is too high. Information is not costless, and the cost or difficulty of obtaining site-specific information may be prohibitive to the producer.

3. Complexity of the proposed production system is too great. There is an inverse relationship between the complexity of a practice and adoption rate.

4. Practice is too expensive. If adoption costs are high in terms of capital outlays and reduced margins, then producers will not be in an economic position to adopt the practice, even if water quality protection is an important goal.

5. Labor requirements are excessive. If a practice requires more labor than the farm manager feels is available, then the practice cannot be adopted.

6. Planning horizon is too short. Some producers may have a short planning horizon because of planned retirement or other factors. If the time associated with recouping initial investments, learning costs, or depreciation of new equipment is beyond the operator's planning horizon, then the practice will not be adopted.

7. Supporting infrastructure is lacking. Producers rely on a network of providers of support and services, such as chemical dealers, implement dealers, extension agents, and other producers. An innovative practice may not be part of the traditional support network's knowledge base. A producer could not adopt such a practice without an adequate support network in place.

8. Producer lacks adequate managerial skill. Many of the new production systems rely on increased management skills, particularly IPM, nutrient management, and precision farming. Producers who do not have the necessary management skills will not adopt such practices.

9. Producer has little or no control over adoption decision. In some cases, a producer cannot make a decision to adopt an alternative practice or production system without the input and approval of partners, landlord, or lender. If these other parties are not convinced of the merits of a proposed change, then the practice cannot be adopted.

10. Information about the practice is inconsistent and conflicting. A producer may hear different messages about the impact of a practice on farm profitability, input needs, and water quality. A producer will be reluctant to adopt a practice until the information about it becomes more consistent.

11. Available information is irrelevant. The information available about the performance of a practice may be based on performance in another county or even another State. A producer may be unwilling to adopt a new practice until information about the practice under local conditions is developed, especially if the new practice entails some investments or changes that are essentially irreversible.

12. Current production goals and new technology conflict. A new technology may not fit into existing production systems or policy settings. For instance, participating in the commodity programs may restrict the ability of a producer to incorporate rotations into his or her operation. A producer may be unwilling to adapt his current operation to fit a new practice.

13. The practice is inappropriate for the physical setting. A practice that was developed for one particular setting, such as flat fertile fields in the Midwest, may cause yield losses, reductions in net returns, or even environmental damage when applied in another setting. A producer will be unwilling to adopt a practice that is inappropriate for his or her setting.

14. Practice increases risk. A new practice may increase the variability of returns. An increase in the risk of a negative outcome may be unacceptable to producers who are risk-averse.

15. Belief in traditional practices outweighs new technology. Some producers are unwilling to abandon practices that are "tried and true," and are therefore perceived as being less risky.

tices. Thus, WQIP payments may be insufficient for adopting and maintaining practices beyond the 3 years that incentives are provided.

The ERS results are supported by a Cornbelt survey (Kraft, Lant, and Gillman, 1996) in which only 17.5 percent of farmers indicated they would be interested in enrolling in WQIP. An additional 27.8 percent stated they might be interested. The average payment requested by those expressing some interest in the program was almost $76 per acre, much greater than the WQIP maximum of $25. Only 18.8 percent were willing to accept $25 per acre or less.

Practice subsidies have also been found to increase the adoption of alternative management practices. Ervin and Ervin (1982) found that government cost-sharing was a significant variable for explaining soil conservation efforts in one Missouri county. Similarly, Nielsen, Miranowski, and Morehart (1989) studied aggregate soil conservation investments and found that cost-shares were a significant variable when conservation tillage was included as an investment. It is important to note that soil-conserving practices produce water quality benefits only as an indirect effect. These practices are designed primarily to enhance long-term soil productivity, which is of immediate economic concern to farmers.

### *Entry and Exit Subsidies: Land Retirement*

The USDA Conservation Reserve Program (CRP) uses subsidies to retire cropland especially prone to producing environmental problems. In exchange for retiring highly erodible or other environmentally sensitive cropland for 10-15 years, CRP participants are provided with an annual per-acre rent and half the cost of establishing a permanent land cover (usually grass or trees). Payments are provided for as long as the land is kept out of production. These subsidies ensure a degree of extramarginal efficiency (i.e., that entry/exit issues are considered to some degree).

CRP eligibility has been based on soil erosion (first 9 signups) and potential environmental benefits (signups 10 and up). With the 10th signup, the cost effectiveness of CRP outlays was increased by using an environmental benefits index (EBI) to target funds to more environmentally sensitive areas. The EBI measures the potential contribution of enrollment bids to conservation and environmental program goals. The seven coequal EBI components are surface-water quality improvement, groundwater quality improvements, preservation of soil productivity, assistance to farmers most affected by conservation compliance, encouragement of tree planting, enrollment in Hydrologic Unit Area Projects of USDA's Water Quality Program, and enrollment in established conservation priority areas. Enrollment bids with a higher EBI to rental payment ratio were accepted ahead of bids with lower ratios. Thus, to some degree, the EBI ensures that land with characteristics most related to environmental quality is enrolled first.

The CRP has converted a total of 36.4 million acres of cropland to conservation uses since 1985, about 8 percent of U.S. cropland. Net social benefits of the CRP are estimated at $4.2-$9 billion (Hrubovcak, LeBlanc, and Eakin, 1995).

## Compliance Mechanisms

Instead of offering farmers a payment to adopt alternative practices, existing program benefits can be withheld unless the change is made. So-called compliance mechanisms tie receipt of benefits from unrelated programs to some level of environmental performance. Examples include USDA's Conservation Compliance program to reduce soil erosion and the Swampbuster program to discourage the drainage of wetlands (USDA, ERS, 1994). As applied to agricultural nonpoint-source pollution, program benefits could be withheld if a conservation or water quality plan containing the appropriate technologies is not developed and implemented. Producers would have an incentive to develop the plan as long as the expected program benefits outweighed the costs of implementing the plan.

The effectiveness of compliance mechanisms for controlling agricultural nonpoint-source pollution is limited by the extent to which those receiving program benefits are contributing to water quality problems. In addition, the effectiveness of a compliance approach varies with economic conditions. Generally, program benefits decrease when crop prices are high. It is precisely during these times that agriculture's pressures on the environment are greatest and the incentive effects of compliance are at their lowest. Budgetary reasons may also force the reduction of program benefits, reducing the incentive effect of compliance mechanisms.[12]

[12] The Federal Agriculture Improvement and Reform (FAIR) Act of 1996 reduces commodity support programs through 2003.

The compliance approach's cost effectiveness depends on how the policy is designed. If the policy requires that particular practices be adopted, then cost effectiveness would be poor if it is not possible to choose the practices optimally. If compliance is based on performance, then producers have an incentive to find the least-cost approach to meeting the performance requirements. However, compliance cannot generally allocate pollution control among farms in a least-cost way because program incentives are unlikely to be distributed in a way that reflects contributions to water quality damages (farms with high damages receiving more program benefits). The administration and enforcement costs for compliance may be high. Individual water quality plans must be developed, and farm-level monitoring and enforcement carried out.

The Food Security Act of 1985 enacted conservation compliance provisions for the purpose of reducing soil erosion. The provisions require producers of program crops who farm highly erodible land (HEL) to implement a soil conservation plan. Reducing soil erosion has implications for water quality. Violation of the plan would result in the loss of price support, loan rate, disaster relief, CRP, and FmHA benefits.

The 1996 NRCS Status Review (USDA, NRCS, 1996) determined that only 3 percent of the nearly 2.7 million fields required to have a conservation compliance plan were not in compliance. USDA estimates that nearly 95 percent have an approved conservation system in place. An additional 3.8 percent are following an approved conservation plan with a variance granted on the basis of hardship, climate, or determination of minimal effect. These results indicate that farmers had sufficient incentives to develop and adopt alternative conservation practices.

Evaluations of conservation compliance report minimal or moderate increases in crop production costs and significant reductions in soil erosion (Thompson and others, 1989; Dicks, 1986), although regional assessments show significant variation in costs and benefits. Two studies conclude that conservation compliance is a win-win situation with increased farm income and reduced soil loss (Osborn and Setia, 1988; Prato and Wu, 1991). However, others show reductions in soil loss are achieved only with decreases in net farm income (Hickman, Rowell, and Williams, 1989; Nelson and Seitz, 1979; Lee, Lacewell, and Richardson, 1991; Richardson et al., 1989; Hoag and Holloway, 1991; Young, Walker, and Kanjo, 1991). The majority of HEL can apparently be brought into compliance without a significant economic burden. A national survey of producers subject to compliance found that 73 percent expected compliance would not decrease their earnings (Esseks and Kraft, 1993).

Conservation compliance has resulted in significant reductions in soil erosion. Annual soil losses on HEL cropland have been reduced by nearly 900 million tons (USDA, NRCS, 1996). Average soil erosion rates on over 50 million HEL acres have been reduced to "T," or the rate at which soil can erode without harming the long-term productivity of the soil. If conservation plans were fully applied on all HEL acreage, the average soil erosion rate would drop from 16.8 tons per acre per year to 5.8 tons (USDA, NRCS, 1996).

Finally, conservation compliance has been calculated to result in a large social dividend, primarily due to offsite benefits. An evaluation using 1994 HEL data indicates the national benefit/cost ratio for compliance is greater than 2 to 1 (although the ratios vary widely across regions) (USDA, ERS, 1994). In other words, the monetary benefits associated with air/water quality and productivity outweigh the costs to government and producers by at least 2 to 1. Average annual water quality benefits from conservation compliance were estimated to be about $13.80 per acre (USDA, ERS, 1994). However, these findings do not necessarily indicate that existing compliance programs are cost-effective nonpoint pollution-control mechanisms.

## Market Mechanisms

The creation of markets for pollution allowances is an innovative approach to reducing pollution from sources with different marginal costs of control. For point sources of pollution, a simple market works as follows. Each source is provided with a permit defining the level of emissions it may discharge, where aggregate allowable emissions for the watershed are determined based on some policy goal.[13] A market is then created

[13] Permits may be allocated to polluters in a number of ways. They may be auctioned or sold to polluting firms by the government, or distributed free of charge on any basis that is deemed fair. The implicit assumption when firms must pay for permits is that they do not hold the right to pollute. When permits are provided free of charge, initial property rights reside with polluters. The initial allocation does not affect the final outcome, only the distribution of wealth.

by letting firms redistribute emissions levels among themselves by buying or selling "allowances," which are essentially authorizations to increase emissions. For example, if firm A purchases an allowance from firm B, then firm A can increase its emissions by the amount specified by the allowance and firm B must decrease its emissions by the same level.

Firms with initial emission levels greater than their initial permit holdings will have to either purchase more allowances or reduce emissions, depending on the relative cost of each method. Firms with higher marginal costs of emissions reduction will purchase allowances from firms with a lower marginal cost of emissions reduction. This sort of trading scheme makes it beneficial for firms with lower pollution control costs to reduce emissions by more than firms with higher control costs, reducing pollution control costs for the watershed as a whole. Point-source allowance markets have been used for a number of years with varying degrees of success. Most successful has been the market for $SO_2$ emissions allowances, which has significantly reduced firms' compliance costs for meeting air quality regulations (USGAO, 1997).

## Permit Markets Involving Nonpoint Sources

A market could be designed to include nonpoint sources. In such a program, point sources would have the option of purchasing allowances from nonpoint sources to meet their emissions reductions requirements. Trading between point sources and nonpoint sources is possible when the pollutants are common to both point and nonpoint sources (e.g., nitrogen and phosphorus), or when the effects of pollutants on expected damages can be used to determine appropriate trading ratios between different types of pollutants. Costs of reducing agricultural nonpoint-source loads in a watershed may be less than reducing point-source loads, especially where point-source discharges are already being constrained by the National Pollution Discharge Elimination System (NPDES) permits of the Clean Water Act.

Point/nonpoint trading is most feasible when both point and nonpoint sources contribute significantly to total pollutant loads (Bartfeld, 1993). If the nonpoint source contributions are very large in relation to the point-source contributions, then the point sources will be unable to purchase enough nonpoint-source allowances to make much difference in water quality. On the other hand, if point sources are very large in relation to the nonpoint sources, savings from trading may not justify the administrative expense of a trading program.

However, point/nonpoint trading is not suitable for all types of water bodies (Bartfeld, 1993). Trading is most suitable for water bodies with long pollutant residence times, such as lakes and estuaries. In water bodies with short pollutant residence times, water quality impacts of nonpoint-source pollution vary with flow levels. During wet periods when nonpoint-source discharges are greatest, stream flow is also higher, and the impacts of nonpoint-source pollutants on stream water quality are lessened through dilution. On the other hand, streams will experience little nonpoint-source discharge during dry periods when flow is low. It is during these periods that point-source discharge impacts on water quality are most severe. Trading will do little to protect water quality during these low-flow conditions.

Efficiency of a trading program is increased if nonpoint sources can trade with other nonpoint sources. Trading between nonpoint sources will occur, however, only if there is an enforceable cap on runoff (or expected runoff). Otherwise, producers would have no incentive to purchase pollution allowances. As with all pollution control policies, trading will be effective only if policy goals represent an improvement over current situations.

### *Choice of Permit Base for Nonpoint Sources*

As with other incentives, the characteristics of nonpoint pollution make it difficult to establish effective markets for nonpoint pollution allowances. Allowances for nonpoint emissions cannot be directly traded because these emissions cannot be measured (Letson, Crutchfield, and Malik, 1993). Even if emissions permits were allocated to nonpoint sources, there would be no way of knowing whether a source was in compliance.

Nonpoint permits provide producers with incentives to reduce pollution. Therefore, as we have shown throughout this chapter, permits can be applied to a number of bases. In this section, we consider two types of permit markets. The first market is defined by point-source polluters trading emissions allowances for allowances based on expected runoff by nonpoint polluters. The second market is defined by point-source polluters trading emissions allowances for allowances based on input use by nonpoint sources. In both cases, allocative efficiency is increased by allowing trades to occur among like sources.

No matter which base is chosen, nonpoint allowances will not generally be traded for point-source allowances one-for-one due to the different allowance bases, the random nature of nonpoint pollution, and the heterogeneous nature of nonpoint-source contributions to pollution. Instead, a trading ratio must be established to define how many nonpoint allowances must be purchased by a point source to equal one unit of emissions allowances, and vice versa.

### *Permit Market Based on Expected Runoff*

A market based on allowances for expected runoff creates the same incentives as taxes/subsidies applied to expected runoff. Under such a system, an efficient, CE(a), or CE(x) outcome will be attainable only under very restrictive conditions (Shortle and others, 1998b). However, a CE(r) outcome is possible in which allowances are traded at a uniform rate. Optimally, agricultural producers would be allowed to trade allowances among themselves, and also with point-source polluters. Expected runoff allowances cannot be traded one-for-one with point-source emissions allowances, however. A uniform trading ratio equal to the price of an emissions allowance relative to the price of an expected runoff allowance defines the number of emissions allowances that must be traded for one unit of expected runoff. As a result of the uniform trading ratio, high social-cost nonpoint polluters will use more inputs than is efficient while low social-cost nonpoint polluters will use fewer inputs than is efficient. Similarly, high social-cost point-source polluters will emit more than is efficient while low social-cost point-source firms will emit less than is efficient.

There are several problems with basing an allowance market on expected runoff. First, monitoring and enforcement costs will be high because the simulation models used to determine compliance require that the technology used and the use of each input be monitored. Second, producers must know how their production decisions affect runoff if the market is to be effective. Government intervention to help ensure that the necessary information is available to producers would likely be expensive. Finally, legal problems may be created if permits are based on the resource management agency's expectations about runoff as opposed to actual runoff, especially given the limited ability of modelers to accurately predict runoff from input use and management practices.

### *Permit Market Based on Input Use*

Shortle and Abler (1997) suggest trading point-source emissions for nonpoint variable production inputs. The efficient trading ratio is defined to be the marginal rate of substitution of emissions for input use such that expected damages and pre-permit profits are held constant (Shortle and Abler, 1997). With $n$ production sites and $m$ inputs that influence pollution, $n$ x $m$ markets (trading ratios) are required to achieve efficiency. Obviously, the transaction costs of such a market system would be considerable (Shortle and others, 1998b).

A second-best allocation could be obtained by allowing trades to occur at uniform rates and by limiting the number of inputs to be traded. The resulting outcome is the same as would occur when uniform input taxes are applied to the same limited set of inputs. The second-best input allowance market economizes on transaction costs associated with monitoring and enforcement of permits for the unrestricted inputs and would reduce the incentives for noncompliance by reducing arbitrage opportunities. Little can be said qualitatively about the second-best prices relative to efficient prices derived by Shortle and Abler (1997). Whether the input allowance prices in the restricted set are higher or lower than their efficient counterparts depends not only on the effects of the input on environmental quality, but also on substitution relationships with other restricted and unrestricted factors.

Uniformity of prices across polluters reduces the cost-effectiveness of pollution control because it eliminates potential gains from different treatment of polluters according to their relative impacts on ambient conditions. The inefficiencies that occur from uniform input prices when differential prices are optimal are analogous to the inefficiencies that can occur when uniform emissions charges are used in place of an optimally differentiated structure (Baumol and Oates, 1988). High control-cost or low social-cost polluters will end up devoting too many resources to pollution control while low control-cost or high social-cost polluters will devote too few resources to pollution control. However, if the differences in the economic gains are small before transaction costs are considered, then even small savings in transaction costs may be justified. If the differences in the gains are large, then the transaction cost savings must be comparably large.

The determination of which inputs are likely to be the best prospects for regulation will depend on the nature of any resulting substitution effects, correlation with environmental quality, and enforcement and monitoring costs. Finally, monitoring and enforcement would be easier for a second-best input market than a market based on expected runoff. Consequently, the costs associated with these activities will probably be less under a market for inputs.

## Empirical Evidence

Point/nonpoint trading programs have been set up to restore water quality in several U.S. water bodies, notably Dillon and Cherry Creek Reservoirs in Colorado, and Tar-Pamlico Basin in North Carolina (Hoag and Hughes-Popp, 1997). These existing programs are designed such that point-source polluters purchase emissions allowances from nonpoint polluters. The amount of allowances purchased depends on the amount of expected runoff to be reduced by nonpoint polluters, and the trading ratio. Under existing programs, expected runoff reductions from nonpoint sources in the basin occur through installation of best-management practices (BMP's) and the development of nutrient management plans. For example, the ratio at which nonpoint expected runoff allowances can be converted to point-source emissions allowances is 2:1 for the Dillon Reservoir, and 3:1 for cropland and 2:1 for livestock for Tar-Pamlico. However, it should be noted that permits were not issued to nonpoint sources.

In several existing programs, the expected cost of reducing nonpoint-source loadings was estimated to be lower than the cost of (further) reducing point-source loadings (table 3-5), suggesting that trades may be beneficial for both parties. However, no trades have occurred (Hoag and Hughes-Popp, 1997). One significant factor may be program design. Because nonpoint sources are not regulated, any trades are not enforceable. Instead, if nonpoint-source reductions failed to meet water quality goals, then point sources would be held responsible for meeting the goal through increased point-source controls. Also, agricultural producers may not have wished to participate for fear of being labeled as polluters and becoming regulated in the future.

**Table 3-5—Estimated marginal phosphorus abatement costs for point and nonpoint sources**

| | Abatement cost | |
|---|---|---|
| Location | Point source | Nonpoint source |
| | *$/pound* | |
| Dillon Reservoir, CO | 860-7,861 | 119 |
| Upper Wicomico River, MD | 16-88 | 0-12 |
| Honey Creek, OH | 0-10 | 0-34 |
| Boone Reservoir, TN | 2-84 | 0-305 |

The range of estimates in each case reflects varying stringency of controls or differences among sources (for example, agricultural versus urban sources).

Source: Malik, Larson, and Ribaudo, 1992.

The Tar-Pamlico program provides good examples of several other problems facing existing point/nonpoint trading programs. The largest point-source polluters in this area formed an association and traded as a group (to reduce transaction costs) at a pre-determined price. Members of the association could purchase nitrogen reduction allowances by contributing to the North Carolina Agricultural Cost Share Program at a fixed price of $56/kg (this price has recently been reduced to $29/kg). The State would then handle the task of getting agricultural producers to participate in the program and deciding how much reduction alternative farming practices would achieve. However, the fixed price was based on average control costs, thus reducing the potential benefits that would have been obtained through margin pricing (Hoag and Hughes-Popp, 1997). Also, the program's requirement of a 2:1 trading ratio may have increased the cost of a trade to levels that have been unattractive to point sources. Initial loading reduction goals for the program were met by the point sources through changes in the production process at a cost of less than $56/kg. Finally, the program is hampered by a lack of generally applicable models or data linking land use practices to water quality effects (Hoag and Hughes-Popp, 1997).

No markets currently exist for trading allowances based on nonpoint inputs. However, literature on second-best input taxation offers some insights into the efficiency loss resulting from the use of uniform prices and trading ratios applied to only a few inputs (see the discussion of input-based incentives in this chapter under the heading, "Applications of Design-Based Incentives").

## Summary

Economic incentives have many desirable characteristics. They rely on market systems to achieve desired outcomes, they allow producers to respond to changes in economic conditions, and (for a given policy objective) they allocate costs of control efficiently among producers by allowing producers to use their own specialized knowledge about their operations.[14] This chapter has focused primarily on the two main classes of incentives: performance-based and design-based. The choice of base is important in determining (1) the types of incentives provided to producers, (2) the degree of flexibility producers retain in their production and pollution control decisions, (3) the complexity of policy design, (4) the informational requirements of both producers and the resource management agency, and (5) the administration and enforcement costs of the policy.

**Instruments perform best when the incentives provided by the instrument coincide with the goals of the resource management agency**. For example, an ambient-based instrument can be designed to achieve a mean ambient goal at least cost (when producers have appropriate expectations about the nonpoint process). However, an ambient-based instrument cannot be designed to achieve an efficient outcome because the incentives provided by the instrument (i.e., to control expected ambient pollution levels) differ from goals of policymakers (i.e., to control expected damages). Likewise, a cost-effective expected runoff-based instrument exists when the objective of policymakers is to achieve a mean runoff goal. However, an expected runoff-based instrument cannot be used to achieve an efficient outcome or to achieve an ambient water quality goal at least cost due to differences in policy goals and incentives provided by the instrument. As another example, suppose nitrogen runoff is a problem in a particular watershed. In this case, incentives applied to fertilizer use and irrigation are likely to be more effective than incentives applied to technology choices that are less correlated with water quality or incentives designed to retire land from production.

**Performance-based instruments can be inferior to design-based instruments on several grounds**. First, runoff-based instruments are not presently feasible because runoff cannot currently be monitored at reasonable cost with current monitoring technology. Second, optimal ambient-based instruments exist only when producers and the resource management agency share the same expectations about the nonpoint process.

Third, the informational requirements for both the resource management agency and producers are increased with ambient-based instruments relative to design-based instruments. For example, producers must be able to evaluate how their actions and the actions of others influence the incentive base for ambient-based instruments to be effective. Moreover, producers have to make predictions about the actions of other polluters before they can predict how their own actions will influence the incentive base. Similarly, the resource management agency must understand how producers will evaluate the incentives. Thus, the agency is required to know what information is available to each producer and how each producer will evaluate that information. Neither producers nor a resource management agency are likely to be able to obtain and process such large amounts of information, which are not required with design-based instruments.

Finally, ambient-based instruments will be less effective if producers are risk averse. In this case, efficiency can be increased if these performance-based instruments are combined with design-based instruments.

**Of the two types of design-based instruments described (i.e., instruments based on expected runoff and instruments based directly on input use and technology adoption), second-best input- and technology-based incentives are most conducive to meeting specified policy goals**. Ideally, instruments should be applied to all inputs and technologies used and be site-specific. However, empirical evidence suggests only a slight welfare loss from using uniform policies applied to only a few key inputs and technologies. The degree of uniformity, the inputs and technologies targeted, and the amount of site-specific information utilized in policy design that provides the best level of control at lowest welfare and administration cost is conditional on local setting, availability of information, and the skill of the resource management agency. Input and technology incentives may be constructed to perform relatively well in promoting least-cost control when the tax or subsidy is closely correlated to pollution control performance (Russell, 1986). For example, if fertilizer application rates are closely correlated with nutrient loadings to a stream because

[14] Economic incentive policies also create incentives for research into more efficient technologies. This is discussed in chapter 7.

of local geographic and hydrologic conditions, then a tax on fertilizer application will achieve a level of control almost as efficiently as a tax on nutrient loadings (Russell, 1986).

In contrast, expected runoff-based instruments are likely to be more costly to administer than other design-based instruments because the resource management agency has to monitor input use and technology choices for each production site and develop a model to predict runoff from all sites.

**Regardless of the choice of instrument base, economic efficiency is increased when additional instruments are used to limit the scale of production in the region**. Otherwise, the mix of production sites will be suboptimal, resulting in too much pollution. Optimal policies would ensure that an optimal mix of land remains in production. However, determining the optimal mix involves a comparison of each site's private net returns to the site's contribution to external social costs, an impractical process when there are a large number of agricultural production sites. Instead, second-best principles can be used to limit the costs of such policies. As with the CRP, the resource management agency may develop alternative criteria on which to limit production, such as identifying extramarginal land on the basis of resource characteristics. For example, land consisting of poor soils, steep slopes, or sandy soils overlying ground water used for drinking water, or land that is close to reservoirs might be identified as extramarginal in the sense that the management practices necessary to reduce the risk of water quality damages to acceptable levels would be prohibitive. Such cropland could be retired through a number of mechanisms, including lump sum taxes, subsidies, regulation, or long-term easements.

**Coordination of existing programs and improved targeting of incentives will lead to further water quality improvements**. Design-based subsidies are being used by USDA and States to promote the adoption of management practices believed to protect water quality. One drawback is that these subsidies are not designed to affect the long-term profitability of a practice. As a result, evidence suggests that they have not successfully promoted the long-term adoption of practices believed necessary to meet water quality goals. A subsidy-based policy could be strengthened by offering long-term subsidies that increase net returns. Another drawback is the technology-based focus of these incentives. While input use may be altered as an indirect effect of adopting alternative practices or technologies, programs will be more successful if incentives are applied directly to input use when this use is highly correlated to water quality impairment.

A final drawback of a subsidy-based policy is that it encourages increases in the scale of production (i.e., production on extramarginal acreage), resulting in more pollution. A separate policy instrument may be required to decrease the scale of production and increase relative efficiency. A lump sum payment or subsidy to retire marginal cropland could achieve this control. (A lump sum tax could also achieve this goal, but such a tax carries the same political baggage as a design tax.) Such a payment is similar to the current CRP, which retires marginal cropland in order to achieve environmental benefits. Coordinating a CRP-like program with long-term incentive programs targeted at both technologies and input use could provide more cost-effective control of nonpoint-source pollution in sensitive watersheds than current programs.

Properly designed market-based systems may be effective alternatives to existing incentive programs. Market-based systems would reduce overall pollution control costs by combining point-source and nonpoint-source policies and allowing markets to allocate pollution control costs more efficiently. The two types of market-based systems that seem to offer the greatest potential are those based on expected runoff and those based on input use. Which type of system performs better is an empirical issue. However, the principles from second-best design incentives may be used in the construction of markets for polluting inputs. A market based on a limited number of inputs may minimize administration costs and still achieve significant pollution control if the inputs are highly correlated with water quality impairments.

The current institutional setting makes point/nonpoint trading difficult and does not favor the establishment of nonpoint/nonpoint trading. A necessary component of a trading program is that the activity the permits are based on (emissions or inputs) can be regulated. Regulations, in the form of emissions permits authorized under the Clean Water Act, exist for point sources. However, nonpoint sources are currently exempt from any regulations. Binding constraints must be imposed on the permitted activities through an enforceable permit system if the market is to operate effectively.

## Appendix 3A—
## Illustration of Some Results

**Proposition 1**. An ambient-based incentive can be designed to achieve the cost-effective solution based on a mean ambient target only when producers and the resource management agency share the same expectations about the nonpoint process.

**Proof**. Denote a producer's site-specific joint distribution function defined over all random variables as $h_i$ (*v*, *W*) where *v* is an (*n*x1) vector with *i*th element $v_i$. In general, a producer's site-specific joint distribution, $h_i$ (*v*, *W*), differs from the resource management agency's, denoted by *g* (*v*, *W*).

Denote the site-specific ambient tax rate by $t_i$. Assuming producers to be risk-neutral, each producer will choose input use to maximize expected after-tax profit, restricted on the choice of technology:

$$V_i(A_i) = \underset{x_{ij}}{Max}\{\pi_i(x_i, A_i) - t_i E_i\{a\}\}$$

where $E_i$ is the mean operator corresponding to $h_i$ (*v*, *W*). The first-order necessary condition for an interior solution is

$$\frac{\partial \pi_i}{\partial x_{ij}} - t_i E_i\{\frac{\partial a}{\partial r_i}\frac{\partial r_i}{\partial x_{ij}}\} = 0 \quad \forall i, j \tag{3A-1}$$

Comparison of (3A-1) with (2B-4) implies the following condition must hold in the optimal solution:

$$t_i E_i\{\frac{\partial a}{\partial r_i}\frac{\partial r_i}{\partial x_{ij}}\} = \lambda * E\{\frac{\partial a *}{\partial r_i}\frac{\partial r_i^*}{\partial x_{ij}}\} \quad \forall i, j \tag{3A-2}$$

where the superscript (*) denotes that these variables are set at their optimal levels in the cost-effective solution. Further manipulation of (3A-2) yields the condition

$$t_i = \frac{\lambda * E\{\frac{\partial a *}{\partial r_i}\frac{\partial r_i^*}{\partial x_{ij}}\}}{E_i\{\frac{\partial a}{\partial r_i}\frac{\partial r_i}{\partial x_{ij}}\}} \quad \forall i, j \tag{3A-3}$$

In general, equation (3A-3) is overdetermined with *m* equations and one unknown. An optimal tax rate exists only when either (1) producers have a single production choice that influences runoff, or (2) $h_i$ (ν, *W*) = *g*(ν, *W*) $\forall i,j$.

**Proposition 2**. A cost-effective expected runoff incentive tax will result in too few sites in production.

**Proof**. The optimal tax rate is $\lambda_i^*$, where $\lambda_i^*$ is defined as the value of $\lambda_i$ in the solution to equations (2B-8) and (2B-9). When faced with an optimal expected runoff tax, the after-tax profits associated with production on the *i*th site are

$$\pi_i(x_i, A_i) - \lambda_i^* E\{r_i\}$$

In a competitive market, production will occur on a site as long as after-tax profits are positive, i.e., as long as

$$\pi_i(x_i, A_i) - \lambda_i^* E\{r_i\} \geq 0$$

The marginal site, *n*, is the site for which after-tax profits vanish, i.e.,

$$\pi_n(x_n, A_n) - \lambda_n^* E\{r_n\} = 0 \tag{3A-4}$$

In general, $n \neq n^*$ where $n^*$ is the solution to (2B-8) and (2B-9) unless (2B-9) is satisfied. Assuming constraint (2B-3) is binding, condition (2B-9) requires that $\pi_{n^*} = 0$, which generally differs from (3A-4), which implies $\pi_n (x_n, A_n) = \lambda_n^* E\{r_n\} > 0$. Therefore, the number of production sites will be too small. An additional instrument is needed to ensure optimal entry and exit. A lump sum refund of the total tax bill would be sufficient to satisfy (2B-9).

# Chapter 4

# Standards

*Standards legally require or mandate that producers behave in a specified manner. Policymakers use standards to control nonpoint pollution by mandating that producers act in a more environmentally conscious manner. In this chapter, we detail a variety of standards that may be used for nonpoint pollution control and evaluate them according to several criteria related to instrument design and implementation.*

## Introduction and Overview

Economic incentives use the price system to get producers to take into account externalities such as polluted runoff. An alternative approach is to legally require or mandate that producers behave in a specified manner. For example, producers may be required to limit input use to a specified level, or they may be required to adopt a specific technology. Behavioral mandates are traditionally referred to as command and control regulations or standards. Traditional water quality policies in the United States, aimed primarily at point sources, have relied on standards.

Standards can be applied either to producers' actions (design standards) or to the results of their actions (performance standards). For point sources, the preferred basis for choosing a standard is emissions because emissions are closely tied to damages and are easy to measure (Baumol and Oates, 1988). However, the choice is not so clear for nonpoint sources, where runoff and other physical processes are difficult or even impossible to observe.

In this chapter, we detail two classes of incentive bases that may be used for nonpoint pollution control:
(1) performance-based standards (i.e., standards in the form of runoff or ambient concentrations), and
(2) design-based standards (i.e., standards in the form of restrictions on inputs and technology). We discuss the major characteristics of and policymakers' experience with a variety of specific standards (table 4-1 lists the standards that are covered in this chapter and provides examples of actual applications of each). Specifically, the optimal form of each standard is developed and evaluated according to (1) its relative efficiency, (2) its relative complexity, (3) informational requirements of regulators in designing the standard and of producers in using the standard to evaluate their decisions, (4) the flexibility of the standard to changing economic and environmental conditions, and (5) potential administration and enforcement costs.

## Performance Standards

Performance standards consist of regulations placed on observable outcomes of a polluter's decisions. For point sources, performance standards are placed on the amounts of pollutants in the effluent leaving the plant. Such discharges are easy to observe and to monitor. The situation is more complex for agricultural nonpoint pollution, however. Agricultural performance bases (i.e., runoff, ambient pollution levels, or damages) cannot be controlled deterministically (without randomness) due to the natural variability associated with the nonpoint process. Therefore, agricultural performance-based standards must be defined in terms of the probability of attainment. For example, consider a standard based on runoff. The standard could be defined in terms of the mean or variance (or other moments) of runoff levels, or it could be defined in terms of a probability (e.g., runoff must not exceed a target level more than 95 percent of the time).

Performance-based standards have several drawbacks. Monitoring would have to occur over a period of time to determine the sample distribution of the base. For example, suppose the standard requires that a producer's mean monthly runoff levels are no greater than $z$. In this situation, it would not be appropriate to take a single monthly measurement and determine a producer to be noncompliant if actual runoff levels are greater than $z$. Instead, measurements must take place over a

**Table 4-1—Types of standards and examples**

| Standards | Actual applications |
|---|---|
| Performance-based: | |
| Runoff | None in existence |
| Ambient | None in existence |
| Design-based: | |
| Inputs | Pesticide label rates; nutrient control laws in several States |
| Technology | Water quality protection laws in a number of States; Coastal Zone Act Reauthorization Amendments |
| Expected runoff | Erosion law in Ohio |

number of months to obtain a large enough sample to have a good estimate of mean monthly runoff levels. Only then could a producer be determined to be in or out of compliance. The required timeframe for monitoring may be significantly longer for some pollutants due to long time lags associated with the delivery of the pollutant to a water body. Some agricultural chemicals, such as phosphorus, can build up in the soil. Changes in management may not result in changes in water quality until the chemical stored in the soil is depleted. It may therefore take years to determine if producers are in compliance with an ambient standard.

Neither the resource management agency nor producers can observe runoff, so it is not possible to determine whether or not a producer is in compliance with such a standard. For ambient standards, producers must have perfect information about their own contribution to ambient pollution levels and also the contributions of others for the standard to be effective (because they must be able to predict how their actions will influence ambient pollution levels). In addition, all producers must have identical expectations about random processes. These requirements severely decrease the likelihood of an ambient standard's being an effective policy measure.

In summary, performance standards based on runoff or ambient quality are not feasible policies for controlling nonpoint-source pollution, given current monitoring technology. Fortunately, removing runoff- or ambient-based performance standards from the set of possible policy tools does not necessarily imply a loss of efficiency. Shortle and Dunn (1986) have shown that design-based standards are more efficient than those based on runoff when economic and environmental uncertainty exists.

# Design Standards

Design standards place restrictions on the use of polluting inputs and/or production and pollution control technologies that are consistent with meeting particular environmental goals. A producer's actions, which are inherently observable by a resource management agency, are therefore the basis for compliance as opposed to whether or not an environmental goal is actually achieved.[1] Two subclasses of design-based standards are discussed in this section. The first subclass is based on expected runoff. The second subclass is based directly on inputs and technology.

## Expected-Runoff Standards

Expected runoff is the level of runoff that is expected to result from a producer's production and pollution control decisions (i.e., input use and technology choices). A design standard based on expected runoff differs from a performance standard based on mean runoff levels because compliance under the former is determined by monitoring each producer's input and technology choices, and then using computer models to determine expected runoff levels. Under such a standard, producers are free to choose input levels and technology in the most efficient combinations as long as the standard is achieved. In addition, an expected runoff standard allows producers to make use of any private knowledge they might have about combining inputs and technology, but only to the extent that the private knowledge can be captured by a model. Special knowledge that is not recognized by the model is of no use to the producer.

Important to note is that there may be legal problems with basing standards on the resource management agency's expectations about runoff as opposed to actual runoff, especially given the current limited ability of models to accurately predict runoff from input use and technology choice. A summary of expected runoff-based instruments is presented in table 4-2.

[1] Design standards have played an important role in U.S. water quality policy toward point sources of pollution. The 1972 amendments to the Federal Water Pollution Control Act require all industrial and municipal point sources of water pollution to install "best practicable treatment," "best available treatment," or "best conventional treatment." Implementation of these rules involved defining specific technologies that had to be adopted.

**Table 4-2—Evaluation of expected runoff-based standards**

| Criteria | Efficient (maximize social welfare) | Cost-effective (runoff targets) | Second-best (Uniform standard, imperfect information) |
|---|---|---|---|
| Incentives provided | Instrument does not exist | Good<br><br>Provides incentives for optimal technology adoption. Additional instruments required to ensure optimal entry/exit.<br><br>efficiency. | Fair<br><br>Cost-effective but not efficient. Does not account for heterogeneity in pollution contributions. Additional instruments targeted at technology adoption and entry/exit may increase |
| Overall complexity | N/A | Medium<br><br>Optimally designed instrument is site-specific. Use of model simplifies implementation. | Low<br><br>Optimally designed instrument is uniform across farms. The use of a model simplifies implementation. |
| Information required by producers | N/A | Medium<br><br>Access to same model as resource management agency simplifies producer's understanding of link between farming practices and runoff. | Medium<br><br>Access to same model as resource management agency simplifies producer's understanding of link between farming practices and runoff. |
| Flexibility | N/A | Medium<br><br>Producers are able to respond to changing market conditions, within the constraints imposed by the model. | Medium<br><br>Producers are able to respond to changing market conditions, within the constraints imposed by the model. |
| Administration and enforcement costs | N/A | High<br><br>Input and technology choices of each farm must be determined. | High<br><br>Input and technology choices of each farm must be determined. |

N/A = Not applicable.
Note: These rankings are subjective, based only on theoretical properties as opposed to empirical evidence. A more reliable table would be based on empirical results that compare each type of policy according to a consistent modeling framework that is representative of the nonpoint problem.

### *Incentives Provided*

Any set of runoff standards will lead to a cost-effective solution. A runoff-based, cost-effective solution is one in which producers will endeavor to meet a mean runoff standard at least cost. As long as producers are profit maximizers, this will be their goal when faced with any expected runoff standard. Optimal entry/exit in the sector is ensured by setting the standard at a level such that it is more profitable for extramarginal farms to retire land from production.

The relative efficiency of the outcome depends on what standards are set. As illustrated in appendix 2B, the use of expected runoff standards will lead to an efficient outcome or an outcome that achieves a mean ambient pollution goal at least cost only under highly restrictive conditions (Horan 1998).[2] Even so, a target that better reflects a site's contribution to expected damages will be more efficient than one that does not.

Applying uniform standards to all farms is a relatively inefficient method of controlling nonpoint pollution. Given that a resource management agency would have to construct a model of each site to determine compli-

[2] Specifically, an efficient standard exists when either (1) the producer makes only a single decision that influences runoff or (2) the covariance between marginal damages and marginal runoff levels is zero for each input (Horan, 1998).

ance, the cost savings of a uniform approach would likely be minimal. Failing to tailor standards to site-specific or regional circumstances results in poor allocative efficiency. As with uniform taxes, uniform standards result in high- (low-) damage-cost farms using more (less) of each pollution-increasing input than is efficient and less (more) of each pollution-decreasing input than is efficient. In addition, uniform standards may not limit acreage in production in the region. Thus, uniform standards do not provide for the efficient scale of the sector. Failure to impose additional standards or other instruments on producers operating on extramarginal acreage further compromises efficiency.

An expected runoff standard will be effective only if producers understand how their production and pollution control decisions will influence expected runoff. The resource management agency may provide producers with this information by giving them access to runoff models that are used for determining compliance. Note that heterogeneous expectations are not a concern here as they are for performance-based standards because compliance is determined using the resource management agency's expectations. There would be no benefit to producers from using their own expectations.

### *Relative Complexity of the Standard*

An expected runoff standard is administratively complex because input use and technology choices must be monitored for each site to determine expected runoff levels (using a model). In addition, producers have to understand how their production and pollution control decisions influence runoff from their farms.

### *Informational Requirements*

The resource management agency requires no special information to set cost-effective standards since the mean runoff target is specified exogenously (i.e., the mean runoff target is not based on any sort of cost-benefit analysis). The resource management agency also requires information on technology and input use from each farm so that runoff can be estimated with a model. The resource management agency's informational requirements are decreased only slightly when a uniform expected runoff standard is used. Only a single standard needs to be set, rather than a standard for each farm, but information from each farm is still necessary to determine whether the expected runoff standard is being met.

Finally, each producer would have to know how his/her production decisions affect runoff if the instrument is to be effective. Information on the relationship between runoff and production decisions may be provided to each producer by the resource management agency.

### *Flexibility Provided by Standard*

An expected runoff-based standard is moderately flexible. Producers are not restricted in how they meet the standard and have some flexibility in adapting to changing economic conditions. However, their ability to take full advantage of their special knowledge is limited by the sophistication of the models being used to predict expected runoff. Compliance is based on the model predictions.

### *Administration and Enforcement Costs*

Administration, monitoring, and enforcement costs are high for expected runoff standards due to their site-specific nature and because the use of each input and technology by each producer must be monitored to determine (through the use of a model) expected runoff. Costs may be only slightly reduced if uniform standards are implemented, as the expected runoff model must still be applied to each farm to determine whether the standard is being met. Finally, any government assistance to ensure that producers have information about runoff relationships for their farm would likely be expensive.

## Input- and Technology-Based Standards

The second subclass of design standards is based more directly on inputs (e.g., levels and forms of agricultural chemicals) and technology (e.g., erosion and runoff controls, irrigation equipment, and collection and use of animal waste). Currently, agricultural design standards have limited use at both the Federal and State levels. Common standards include pesticide use restrictions and bans, the design of animal waste storage lagoons for large concentrated animal feeding operations, and use of nutrient management practices in areas where drinking water is threatened by polluted runoff.

A summary of input- and technology-based standards (not including expected runoff-based standards) is presented in table 4-3.

**Table 4-3—An evaluation of design-based standards**

| Evaluative criteria | Efficient or cost-effective (maximize social welfare or runoff target) | Second-best (uniform, limited set of inputs, imperfect information) |
|---|---|---|
| Incentives provided | Good | Fair |
| | Provides incentives for optimal input use, optimal technology adoption, and efficient entry/exit | Not efficient. Additional instruments may be required to ensure optimal technology adoption and optimal entry/exit. |
| Overall complexity | High | Low |
| | Standards are site-specific, and must be set for each input and technology. | Standards set for few inputs or are uniform across fields. |
| Information required by producers | Low | Low |
| | No special information required | No special information required. |
| Flexibility | Low | Low |
| | Regulator must change standards as prices change or new technologies are introduced. | Regulator must change standards as prices change or new technologies are introduced. |
| Administration and enforcement costs | High | Medium |
| | Use of each input and technology choice must be monitored. | Use of easily observed inputs must be monitored. |

Note: These rankings are subjective, based only on theoretical properties as opposed to empirical evidence. A more reliable table would be based on empirical results that compare each type of policy according to a consistent modeling framework that is representative of the nonpoint problem.

### *Incentives Provided*

Input and technology subsidies can be designed to achieve an efficient or (any type of) cost-effective outcome (i.e., an outcome that achieves a mean ambient water quality or runoff goal at least cost. See table 2-1). The reason is that input and technology choices, while not equivalent to specific policy goals, are the means by which a resource management agency can achieve its goals. For example, if a resource management agency had absolute control over farm production in a region and wanted to achieve an efficient outcome, it could achieve that outcome by choosing "correct" input use and technologies for the region.

Instruments must target all inputs and technology choices to attain an efficient or cost-effective outcome. Assuming a competitive agricultural sector with no market distortions, ex ante efficient standards would require each producer to employ the efficient site-specific technology and input levels characterized by the three efficiency conditions ((2A-1), (2A-2), and (2A-4)) in appendix 2A. Similarly, cost-effective standards would be the solution to the optimality conditions derived in appendix 2B. Efficient or cost-effective standards are site specific due to land heterogeneity. For example, identical fertilizer application rates on two fields may result in different discharges to surface water because of differences in topography and vegetation between fields and water resources. In addition, standards must be applied to each input that influences pollution, including those that are not currently being used. Input standards typically represent a maximum level of input use that is allowed by law. However, for inputs that reduce runoff, input standards must be defined as the minimum level of input use allowed.

Using standards to control technology is more straightforward than using incentives because the technology choice is mandated as opposed to induced. As a result, the choice of technology in the following discussion is trivial. The resource management agency chooses the technology that yields the greatest level of expected net benefits for society under the framework imposed (i.e., efficient or second-best).

Finally, the efficient scale of production in the industry is guaranteed by setting technology and input standards for production on extramarginal land at levels to prevent profitable operation on this land.

Policies may be designed optimally even when producers retain private information. The resource management agency may have imperfect information about production practices, land productivity, and other site-specific characteristics that affect runoff or economic returns, and producers may be reluctant to truthfully reveal any private information they possess. The resource management agency may therefore have to design a second-best benchmark that does not require obtaining producers' private information.[3] Optimal standards would be the solution to such a benchmark.[4]

Without considering administration and enforcement costs, policy designed with limited site-specific information will generally be less efficient than policy designed under perfect information. However, given the large costs of obtaining site-specific information, policy designed without the benefit of producers' private information may actually be preferred.

Political or legal reasons or costs may limit the ability of a resource management agency to implement site-specific standards for each input that contributes to pollution. Instead, standards may be applied uniformly across sites and applied to only a few inputs, generally reducing administration costs. Inputs to target could be based on ease of observation or measurement. Some management practices, such as the rate at which chemicals are applied, are very difficult to observe without intensive and obtrusive monitoring.

As with incentives applied to a limited number of inputs, optimal standards must be designed to account for input substitution (see appendix 4A). Placing standards on the most easily observed inputs can lead to substitution distortions and undesirable changes in the input mix (Eiswerth, 1993; Stephenson, Kerns, and Shabman, 1996). For example, a standard on herbicides would reduce herbicide use, but may increase mechanical cultivation and soil erosion, which in turn impairs water quality. The resource management agency would have to carefully consider the management alternatives to the undesirable practices, and have in place other measures to counter any undesirable characteristics of the alternatives.

Failing to tailor standards to site-specific circumstances results in poor allocative efficiency. The resource management agency cannot easily target low-cost pollution abaters, and therefore cannot efficiently allocate pollution control efforts to minimize abatement costs. As with uniform taxes, uniform standards result in high (low) damage-cost farms using more (less) of each pollution-increasing input than is efficient and less (more) of each pollution-decreasing input than is efficient. However, unlike the case of uniform input taxes, marginal per acre profits are not equated across farms under uniform standards. In addition, uniform standards may not limit the acreage in production in the region. Thus, uniform standards do not provide for the efficient scale of the sector. Failure to impose additional standards or other instruments on producers producing on extramarginal sites further compromises efficiency.

In general, there is a tradeoff between administration costs and allocative efficiency. Nationwide design standards that are easy to observe, to administer, and to enforce can lower administration costs. Gathering information to better target where controls are applied and developing a broader set of design standards that apply to diverse conditions can significantly increase administration costs. Efficiency is improved if local, rather than national, standards are applied.

### *Relative Complexity of the Standard*

Input- and technology-based standards are relatively simple because they are applied directly to the most basic production decisions. However, these standards are administratively complex because each input and technology choice must be monitored for each farm. Other things equal, site-specific standards will be administratively more complex than uniform standards, and standards applied to each input will be more complex to administer than standards applied to

[3] Policies designed under imperfect information cannot be designed to attain a specific outcome. With limited information, the resource management agency can design policy based only on how it expects producers to react. Therefore, policy would have to be designed to attain an expected outcome.

[4] Second-best standards, while having many of the same properties as second-best incentives, will generally result in different outcomes. This is addressed in chapter 8.

only a few inputs. Finally, standards designed with limited information will be less complex from an administrative perspective.

### *Informational Requirements*

The resource management agency must have perfect information about production and runoff functions for each acre of land in production to achieve efficient or cost-effective pollution control. However, second-best policies may be designed with only limited information about site-specific characteristics. Producers have no special informational requirements with (efficient, cost-effective, or second-best) input- and technology-based standards. They simply operate under the constraints imposed by the standards.

### *Flexibility Provided by the Instrument*

Input- and technology-based standards (efficient or second-best) leave producers and administrators with little flexibility in making decisions or in adjusting policies to meet changing economic and environmental conditions. Specifically, producers are constrained by the standard, and all adjustments to changing economic conditions must be made through changes in the use of unrestricted inputs and technologies. Changes in economic conditions require the resource management agency to set new standards if pollution control is to be cost effective.

### *Administration and Enforcement Costs*

Administration, monitoring, and enforcement costs are high for all efficient (or cost-effective) design-based standards due to their site-specific nature and because use of each input and technology must be monitored. Second-best standards are less costly to apply because they do not have to be site-specific, nor does every input and technology choice have to be monitored for each acre of land in production.

## Application of Design-Based Standards

Until recently, standards had only a limited history of application to agricultural nonpoint-source problems. Performance standards have not been applied to nonpoint-source pollution because it cannot be observed. However, design standards are becoming a more important part of nonpoint-source pollution control policies, primarily at the State level. The performance of most of these programs has yet to be evaluated. Some of the examples presented below are empirical studies of hypothetical nonpoint pollution control programs.

### *Input Standards*

Helfand and House (1995), in a study of lettuce production in Salinas Valley, California, determined cost-effective and second-best input standards when only two inputs—nitrogen and water—influence runoff. To achieve a 20-percent reduction in nitrogen runoff, they found that a uniform rollback of both water and nitrogen use resulted in a welfare loss (relative to the cost-effective baseline) only slightly higher than input taxes. A single standard on water or nitrogen use only resulted in a greater welfare loss.

A study of the economic impacts of alternate atrazine control policies concluded that a partial ban, targeted to particular areas to meet Safe Drinking Water Act standards, was more cost effective than a total ban on atrazine (Ribaudo and Bouzaher, 1994). The cost of reducing surface-water exposure to herbicides under the partial ban was about one-fifth the cost per unit under a total ban. Partial bans allow most producers to continue to use the pesticide, thus limiting increased production costs to relatively few producers. Administration and enforcement costs are higher for partial bans.

### *Technology Standards*

Many States have incorporated enforceable mechanisms for agricultural runoff in their water quality policies (table 1-5 in chapter 1). These mechanisms almost always consist of a farm-level management plan built around “acceptable” management practices. In areas where water quality impairments are known to occur, more stringent practices and enforcement are called for. Most of these laws have been passed only recently, and results in terms of reduced runoff, costs to producers, and costs to States have yet to be documented.

### *Design Standards With Triggers*

A program in Nebraska uses design standards in conjunction with performance measures (Bishop, 1994). Increasing concentrations of nitrate in groundwater led to a 1986 law requiring Natural Resource Management Districts (NRD’s) to require best-management practices to protect water quality. The practices required depended on nitrate concentrations in groundwater. In Phase I areas (the least contaminated), fall applications of commercial nitrogen fertilizer are banned on sandy soils. In Phase II areas (12.6-20 ppm nitrate-N concentrations in groundwater), irrigation wells are to be sampled, irrigation applications metered, deep soil analysis for nitrate required on every field, a ban on fall fertilizer applications instituted on sandy soils, and a ban on any application on heavier soils until after November 1. Phase III (greater than 20 ppm) is the same as Phase II, plus all fall and winter fertilizer applications are banned, and spring applications must be split applications or must use an approved inhibitor.[5] This policy approach is similar to a design standard with imperfect information. The NRD does not know *a priori* which set of management practices will achieve the groundwater quality goal. Design standards are instead changed in response to observed changes in groundwater quality.

Monitoring in the Central Platte NRD, which had the greatest problem, has shown a decrease in groundwater nitrate (Bishop, 1994). No economic assessment on the benefits and costs of the policy has been conducted.

Wisconsin's programs for protecting groundwater from pesticides derive from the Wisconsin Groundwater Law (1983) (Wisc. Stats., Chapter 160), which requires the State to undertake remedial and preventive actions when concentration "triggers" are reached in groundwater for substances of public health concern, including a number of pesticides. Two triggers are established for each chemical, an enforcement standard and prevention action limit (PAL). The PAL is 10, 20, or 50 percent of the enforcement standard, depending on the toxicological characteristics of the substances. When a PAL is exceeded, a plan for preventing further degradation is prepared. When the enforcement standard is exceeded, the chemical is prohibited in that area overlaying the contaminated aquifer.

For example, the enforcement standard for atrazine is 3.5 ppb, and the PAL is 0.35 ppb. Well monitoring found atrazine concentrations in many areas of the State above the PAL (Wolf and Nowak, 1996) and in some areas above the enforcement standard. This prompted the passage of the Atrazine Rule, which established maximum atrazine application rates and conditional use restrictions for the State (Wisc. Admin. Code, Agri. Trade & Cons. Prot. Ag30), as well as zones where additional restrictions are imposed on top of the statewide rules. The result is a three-tiered management plan: statewide atrazine restriction, Atrazine Management Areas where concentrations exceed the PAL, and Atrazine Prohibition Areas where concentrations are above the enforcement standard. Statewide atrazine restrictions impose soil-based maximum application rates, restrict when atrazine can be applied, and prohibit applications through irrigation systems. Further restrictions are placed on application rates in the Atrazine Management Areas. In 1993, 6 management areas and 14 prohibition areas had been established (Wolf and Nowak, 1996).

An assessment of the Atrazine Rule reported that producers in the Atrazine Management Areas were not at a disadvantage to producers who were not in such areas, as represented by comparisons of yield loss predictions and assessment of weed intensity (Wolf and Nowak, 1996). However, an assessment of compliance costs was not made.

## Summary

Standards use the regulatory system to mandate that producers adopt more socially efficient production methods. These mandates may leave producers with little freedom when it comes to their production and pollution control choices. This chapter has focused on the two main classes of standards: performance-based and design-based. The choice of base is important in determining (1) the relative efficiency of the standard, (2) the degree of flexibility producers retain in their production and pollution control decisions, (3) the complexity of policy design, (4) the informational requirements of both producers and the resource management agency, and (5) the administration and enforcement costs of the policy.

**The relative efficiency of the standards is greatest when they coincide with or support the goals of the resource management agency.** Expected runoff standards are cost effective because they can always be used to achieve a mean runoff goal at least cost. However, an expected runoff-based instrument cannot be used to achieve an efficient outcome or to achieve an ambient water quality goal at least cost. As another example, suppose nitrogen runoff is a problem in a particular watershed. In this case, standards applied to fertilizer use and irrigation are likely to be more effective than standards that are applied to the type of crop grown.

[5] Soil inhibitors reduce the rate at which nitrogen is converted to the soluble nitrate form, thus reducing losses to leaching or runoff.

**Performance standards can be inferior to design standards on several grounds.** All of the drawbacks for performance-based incentives hold for standards as well, with an additional drawback for performance standards that is probably even more troublesome. Due to the natural variability associated with the nonpoint process, performance-based standards must be defined in terms of a limit on mean ambient pollution or runoff levels or in terms of a probability associated with the occurrence of certain outcomes. As a result, monitoring would have to occur over a period of time to determine the sample distribution of the base. Only then could a producer be determined to be in or out of compliance. The required timeframe for monitoring may be years for some pollutants due to long time lags associated with the delivery of the pollutant to a water body.

**Standards leave producers with little flexibility.** Standards, since they mandate or limit specific actions, leave producers with little flexibility in adapting to a changing economic environment. Expected runoff standards leave producers with the most flexibility because specific production methods are not specified and producers are free to adjust production as economic conditions change (as long as the standard is met). In contrast, standards on inputs and technologies are totally inflexible. Producers can respond to changing economic conditions only by altering the use of inputs and technologies that are not targeted by the standards.

Some flexibility may be imparted by basing standards on environmental triggers. Allowing continued use of a pesticide after it has been detected in groundwater, but at lower rates, is less costly to producers than immediately banning it. Such an approach lessens excessive regulatory burden resulting from the uncertainties of the effectiveness of best management practices in reducing nonpoint-source pollution. This flexibility comes at a cost of greater administration and monitoring costs.

**Second-best input and technology standards are more practical from an implementation standpoint.** Ideally, standards should be applied to all inputs and technologies used, and be site specific. However, empirical evidence suggests only a moderate welfare loss from using uniform policies applied to only a few key inputs and technologies. The degree of uniformity, inputs and technologies targeted, and the amount of site-specific information utilized in policy design that provides the best level of control at lowest welfare and administration cost is an empirical question. These issues will generally depend on the local setting, availability of information, and the skill of the resource management agency.

Input and technology standards may be constructed to perform relatively well in promoting least-cost control when the standard is closely correlated to pollution control (Russell, 1986). For example, if fertilizer application rates are closely correlated with nutrient loadings to a stream because of local geographic and hydrologic conditions, then a standard on fertilizer applications will achieve a level of control almost as efficiently as a standard on nutrient loadings (Russell, 1986).

In contrast, expected runoff standards are likely to be more costly to administer than other design standards because the resource management agency has to monitor input use and technology choices for each production site and develop a model to predict runoff from all sites.

**Broadening the scope of current programs and improved targeting would lead to further water quality improvements.** A limited number of programs now include design standards as a method of improving water quality. These exist primarily in two forms: standards on technologies and bans on hazardous chemical inputs. A chemical ban is probably reasonable for extremely hazardous chemicals being used in environmentally sensitive areas. However, for areas that are less sensitive and for chemicals with limited risk, a more flexible approach may be more efficient. Some States are addressing this issue by using water quality measures to define specific geographic areas where design standards are imposed and environmental triggers within these areas to define the particular set(s) of standards that are required.

While input use may be altered as an indirect effect of mandating alternative practices or technologies, more direct effects may be desired. Programs will be more successful if policies are applied directly to input use when this use is highly correlated to water quality impairment.

# Appendix 4A—
# A Limited Set of Input Standards[6]

For simplicity, suppose standards are site-specific but applied only to a subset of the total number of inputs. Also for simplicity, we do not explicitly consider technology choices. Let $z_i$ denote the $(m' \times 1)$ vector of inputs whose use is standardized, and let $y_i$ denote the $([m - m'] \times 1)$ vector of inputs that are chosen freely by producers (note that $x_i = [y_i\ z_i]$). Each producer faces the following problem for production on each acre:

$$\begin{aligned} &\max_{x_{ij}} \quad \pi_i(y_i, z_i) \\ &s.t. \quad z_{ij} \le \bar{z}_{ij} \quad \forall j \in [1, k] \\ &\qquad z_{ij} \ge \bar{z}_{ij} \quad \forall j \in (k, m'] \end{aligned}$$

where $\bar{z}_{ij}$ represents that standard for the $j$th restricted input. Inputs denoted by $j \in [1, k]$ are assumed to be pollution-increasing while inputs denoted by $j \in [k, m']$ are assumed to be pollution-reducing. The Lagrangian corresponding to the $i$th acre is

$$L_i = \pi(y_i, z_i) + \sum_{j=1}^{m} \lambda_{ij} [\bar{z}_{ij} - z_{ij}]$$

where $\lambda_{ij}$ is the Lagrangian multiplier for the $j$th restricted input used on the $i$th acre. Assuming an interior solution for all inputs and that all constraints are binding, the necessary conditions for a maximum are

$$\frac{\partial \pi_i}{\partial y_{ij}} = 0 \quad \forall i, j \qquad \text{(4A-1)}$$

$$\frac{\partial \pi_i}{\partial z_{ij}} = \lambda_{ij} \quad \forall i, j \qquad \text{(4A-2)}$$

$$\bar{z}_{ij} - z_{ij} = 0 \quad \forall i, j \qquad \text{(4A-3)}$$

Note that $\lambda_{ij} < 0$ for inputs that reduce runoff. Input use on the $i$th acre is determined by the simultaneous solution to $m + m'$ conditions in (4A-1)-(4A-3). Use of (unrestricted) input $j$ will be a function of the standards for all restricted inputs, $y_{ij}(z_i)$, where $\bar{z}_i$ is an $(m' \times 1)$ vector whose $j$th element is $\bar{z}_{ij}$.

For simplicity, assume that producers hold no private information. Optimal input standards are determined by plugging the (unrestricted) input demand functions (i.e., $y_{ij}(\bar{z}_i)$) into the agency's objective function and choosing input standards to maximize expected net benefits, restricted on technology.

$$J(\bar{A}) = \underset{\bar{z}_{ij}, n}{Max} \left\{ \sum_{i=1}^{n} \pi_i(y_i(\bar{z}_i), \bar{z}_i)) - E\{D(a)\} \right\}$$

The first-order conditions are given by (2A-2) and

$$\frac{\partial J}{\partial \bar{z}_{iu}} = \frac{\partial \pi_i}{\partial \bar{z}_{iu}} + \sum_{j=1}^{m-m'} \left[\frac{\partial \pi_i}{\partial y_{ij}}\right] \frac{\partial y_{ij}}{\partial \bar{z}_{iu}}$$

$$-E\{D'(a) \frac{\partial a}{\partial r_i} \left[ \sum_{j=1}^{m-m'} \frac{\partial r_i}{\partial y_{ij}} \frac{\partial y_{ij}}{\partial \bar{z}_{iu}} + \frac{\partial r_i}{\partial \bar{z}_{iu}} \right] \} = 0 \quad \forall i, u$$

Using (4A-1) and (4A-2), condition (4A-4) can be simplified to yield

$$\begin{aligned} \frac{\partial \pi_i}{\partial \bar{z}_{iu}} = \lambda_{iu} &= E\{D'(a) \frac{\partial a}{\partial r_i} \frac{\partial r_i}{\partial \bar{z}_{iu}}\} \\ &+ E\{D'(a) \frac{\partial a}{\partial r_i} \sum_{j=1}^{m-m'} \frac{\partial r_i}{\partial y_{ij}} \frac{\partial y_{ij}}{\partial \bar{z}_{iu}}\} \quad \forall i, u \end{aligned} \qquad \text{(4A-4)}$$

The optimal shadow value for the $u$th restricted input for the $i$th acre is equal to the marginal damage created by use of the $u$th restricted input on that acre, plus an adjustment term to account for the indirect effect on damages resulting from the effect of the standard on the use of other inputs.

[6] The mathematical foundations for input standards, applied to a limited set of inputs, are developed in this appendix. Unless otherwise stated, the underlying model and assumptions are as developed in appendix 2A.

**Chapter 5**

# Liability Rules

*Liability rules are used to guide compensation decisions when polluters are sued for damages in a court of law. Such rules, although employed only after damages occur and only if victims are successful in their suit, can provide ex ante incentives for polluters to use more environmentally friendly production practices. In this chapter, we discuss two relevant types of liability rules.*

## Introduction and Overview

Individuals who are damaged monetarily or otherwise by the activities of others may have the right to sue for damages in a court of law. If the suit is successful, the court may be guided in its compensation decision by a rule of law or precedent, known as a **liability rule**. The liability rule, while imposed *ex post*, serves as an *ex ante* incentive to deter individuals or firms from engaging in activities that may be damaging to others. For example, liability rules can be designed to hold polluters liable for the damages they cause. If polluters feel that their production decisions may result in damages for which they may be held liable, then they will likely weigh the benefits from participating in pollution-related activities against the penalties that they may expect to face as a result of their actions.

In this chapter, we will review how liability creates incentives to influence producer behavior, and the different forms these rules can take. For each form, we discuss the properties of the rule and compare them with other types of incentives.

## Important Features of Liability

Liability rules are a form of performance-based incentive in that they are imposed after damages are realized (Shavell 1987). However, liability rules differ from traditional performance-based incentives because they are imposed only if a suit is privately or publicly initiated, and if a court of law rules in favor of the damaged parties.[1] Instances may therefore arise in which damages occur but no payments are made.

[1] Shavell (1987) discusses circumstances for which publicly and privately initiated approaches are most appropriate.

Liability rules can be developed under two different frameworks that are relevant for polluters: (1) strict liability and (2) negligence. Polluters are held absolutely liable for payment of any damages that occur under **strict liability**. Polluters are liable under a **negligence** rule only if they failed to act with the "due standard of care" (Segerson, 1995). For example, a producer would presumably not be found negligent (and hence liable) in the pesticide contamination of groundwater if the pesticide was applied in accordance with the manufacturer's specification and the laws regarding application procedures.

When multiple polluters exist, the principle of "joint and several liability" allows damage costs to be divided among polluters according to any distribution of the court's choosing (unless a specific distributional rule takes precedent). The distribution does not have to be based on the polluter's marginal contribution to damages. In fact, it is possible that one polluter could be held liable for all damages. That polluter is then free to sue other responsible parties to share the burden (Miceli and Segerson, 1991; Segerson 1995).

The relationship between polluters and the victims is important for choosing an appropriate liability rule. The relationship may be defined as one of either unilateral care or bilateral care (Segerson, 1995). **Unilateral care** is a situation in which only the polluter influences damages. In other words, the victim has no way of protecting himself. Alternatively, it is sometimes possible for the victim to protect himself. For example, the victim may be able to purchase a filtration system to protect against contaminated ground water. This situation is known as **bilateral care**, and any liability rule takes into account the potential for each party to act to reduce damages. Under some rules, liability is not assessed to polluters if the victim failed to take reasonable preventive actions (Segerson 1995).

## Liability When Victims Cannot Protect Themselves (Unilateral Care)

The following discussion is based on the assumption of joint and several liability. In addition, unilateral care is assumed because strict liability rules are efficient if polluters can undertake preventive actions for the victims (Segerson 1990) (e.g., producers could purchase water purifiers for all victims if that is the least-cost solution for efficient pollution abatement).

### Strict Liability Rules

Producers face uncertainty as to whether or not they will successfully be sued for damages resulting from agricultural nonpoint pollution. This uncertainty is likely to be site-specific and to depend on the ambient pollution level that results from the collective actions of all producers, as well as other uncertain factors such as knowledge of pollutant transport and the ability to identify individual pollutant sources. Consequently, each producer has expectations relating to natural events that influence pollution, the probability of successfully being sued, and other uncertain factors that might influence this probability. In general, producers' expectations may differ from those of the resource management agency defining the rules.

A strict liability rule that can be used to attain efficient nonpoint-source pollution control is developed in appendix 5A. The rule is developed so that each producer expects to pay the total expected damages from pollution, plus or minus a lump sum component that distributes payments across polluters so that total payments equal total damages. However, while each producer expects to pay the same variable portion of the liability rule, the actual rule would have to be site-specific to account for each producer's beliefs about the nonpoint process and about the probability of being sued and found liable. Liability must be higher for producers who do not believe they will be sued and/or found liable to achieve optimal pollution control. Effectively, the site-specific aspects of the rule alter the uncertainty each producer faces about random events (weather, economic conditions) and the prospect of being sued and held liable so that the producers' and the resource management agency's expectations about uncertain events are the same. Equivalent expectations is a condition for efficient pollution control.

Finally, lump sum components must be applied to producers operating on extra-marginal land to ensure optimal entry and exit. Unlike other incentive-based instruments such as taxes, it is not possible to use lump sum instruments to reduce producers' payments to zero under liability rules because the victims must be compensated. Therefore, lump sum portions of the liability rule can be applied to producers operating on marginal and inframarginal acreage and designed to ensure that total liability payments equal total damages. This could be accomplished by providing each producer with a refund of the variable liability payment, and dividing total damages among all producers according to some distributional rule.

### Negligence Rules[2]

Under a liability rule based on negligence, a producer is held liable only if he/she failed to operate under the "standards of due care." "**Due care**" can be measured either in terms of performance-based outcomes or in terms of a producer's actions. Producers may collectively be held negligent if realized damages from pollution in a water body are found to be in excess of some acceptable level. Excess damages would be an indication that at least some producers in the watershed are not using acceptable production practices. Under this rule, all producers in a watershed would be liable for damages if affected parties brought suit. Such a negligence rule, however, does not correct for suboptimal entry and exit. Because the rule applies only to those producers operating at the time the damages occurred, there is no mechanism for applying lump sum components to guarantee optimal entry and exit (Miceli and Segerson, 1991). In addition, by producing at suboptimal levels to avoid the possibility of liability, producers may bring into production more than the economically efficient amount of land (Miceli and Segerson, 1991).

Alternatively, an individual producer may be held negligent if inputs that increase runoff are used above optimal levels, inputs that mitigate runoff are used below optimal levels, or if the technology in use per-

[2] The mathematical basis for negligence rules is developed in appendix 5B.

forms poorly in reducing runoff relative to the optimal technology. Damages would be paid only by those producers not using acceptable production practices. This approach would be more costly to administer than the pollution-based rule, since the acceptable management practices would have to be identified for each site, and each site would have to be monitored for compliance. However, it is more fair in that only those producers who are likely generating unacceptable levels of runoff would be liable. In addition, an efficient solution will generally be attainable.

## Liability When Victims Can Protect Themselves (Bilateral Care)

Situations may exist in which victims have opportunities to take precautions that producers cannot take for them (Wetzstein and Centner, 1992). If so, then strict liability rules applied to producers are no longer efficient because victims may suboptimally protect themselves if they feel that they can collect the full amount of damages. This result would apply to the negligence rules derived in appendix 5B as well, since the components of these rules are based on strict liability. Wetzstein and Centner (1992) suggest the use of a modified strict liability rule based on victim precaution requirements. While not derived here, a modified rule as they propose could be incorporated into either of the negligence rules developed in Appendix 5B. For example, negligence rules would be recommended for relatively safe agricultural chemicals, while strict liability would be recommended for the use of more hazardous materials.

## Empirical Evidence

Both State and Federal regulators have tended to hold producers liable for damages resulting from chemical use only if they failed to apply registered chemicals in accordance with the manufacturer's instructions and any related laws (Wetzstein and Centner, 1992; Segerson, 1990; Segerson, 1995). For example, the Comprehensive Environmental Response, Compensation and Liability Act (CERCLA) restricts producers' liability in this manner.

In more than 30 States, agricultural producers applying chemicals that contaminate groundwater may be held liable under a strict liability standard (Centner, 1990). Groundwater exemption legislation that holds producers to a negligence rule has been passed or proposed in Arizona, Connecticut, Georgia, Iowa, Minnesota, New York, and Vermont. Producers in these States would be exempt from strict liability if they use chemicals "properly." In Connecticut, a producer is required to keep records of pesticide use and groundwater protection plans for 20 years after application to demonstrate due care (Lee and Leonard, 1990).

Many States make compliance with acceptable agricultural best-management practices a defense to nuisance actions (ELI, 1997). Negligence rules of this sort are consistent with the philosophy that producers have a basic "right to farm" and that they should not be penalized as long as they adhere to standard, accepted practices. However, because current negligence rules are based on what has been accepted historically, they may not reflect the current damages caused by previous "standard, accepted practices," and pollution levels will be excessive relative to optimal levels.

## Summary

The characteristics of nonpoint-source pollution, including dispersion of harm and the inability to identify sources, could make very small the probability of a producer being sued and held liable under strict liability rules. A negligence rule may be more appropriate in these cases because it is not necessary to prove a producer's contribution to damages. A producer would not be held liable if he/she complied with acceptable farming practices.

In general, liability rules suffer from many of the same problems that ambient-based incentives do. To achieve an optimal solution, all producers must have realistic beliefs about their collective effects on ambient pollution levels, the profit functions for all sites, and the joint distribution functions of all other producers. The rule-making system must account for each producer's beliefs about the actions of other producers and about aspects of the nonpoint process. For negligence rules, the system must also have site-specific information about producers as well as information about the nonpoint process in order to identify the "optimal" set of practices that defines "due care." These unrealistic assumptions about the information required for producers and the rule-making system limit the feasibility of liability rules.

Finally, the litigation process for liability may be expensive relative to other regulatory methods (Shortle and Abler, 1997). This expense may prevent individuals from attempting to claim damages, letting polluters go unregulated (Shavell, 1987). Thus, liability rules are likely to be at most second-best when transaction costs are considered, and are probably best suited for the control of pollution related to the use of hazardous materials or for infrequent occurrences such as accidental chemical spills or manure lagoon breaks (Wetzstein and Centner, 1992; Shortle and Abler, 1997).

## Appendix 5A— Strict Liability Rules[3]

Suppose the extent of producers' liability depends on the damages that arise as a result of the ambient pollution level. It is appropriate for the liability rule to depend on the ambient pollution level as well. Define a site-specific liability rule in general terms by the function $L_i(a)$. Producers are held liable only if they are sued by a damaged party and are found to be responsible. Therefore, producers face additional uncertainty about whether or not they will be held liable. Producers have their own beliefs regarding the site-specific probability that they will be sued and held liable, and their own beliefs about the distribution of random variables influencing natural events. Denote the site-specific probability that a producer will be sued and held liable as $q_i(a, \eta_i)$, where $\eta_i$ is a vector of random variables that may influence this probability.[4] Similarly, denote a producer's site-specific joint distribution function defined over all random variables as $h_i(v, W, \eta)$ where $v$ is an ($n$x1) vector with $i$th element $v_i$, and $\eta$ is an ($n$x1) vector with $i$th element $\eta_i$. In general, a producer's site-specific joint distribution, $h_i(v, W, \eta)$, differs from the rule-making system's, denoted by $g(v, W)$.

Assuming producers to be risk-neutral, each producer will choose input use to maximize expected per-acre profit, restricted on the choice of technology

$$V_i(A_i) = \underset{x_{ij}}{Max}\left\{p_i(x_i, A_i) - E_i\{q_i(a,\eta_i)L_i(a)\}\right\}$$

where $E_i$ is the mean operator corresponding to $h_i(v, W, \eta_i)$. The first-order necessary condition for an interior solution is

$$\frac{\partial p_i}{\partial x_{ij}} - E_i\{[q_i(a,\eta_i)L_i'(a) + \frac{\partial q_i}{\partial a}L_i(a)]\frac{\partial a}{\partial r_i}\frac{\partial r_i}{\partial x_{ij}}\} = 0 \quad \forall i, j \tag{5A-1}$$

The solution to (5A-1) yields input use as a function of technology choice, $x_i(A_i)$. The producer's optimal choice of technology, $A_i^{**}$, will satisfy the following condition

$$\begin{aligned} V_i(A_i^{**}) - V_i(A_i') &= [p_i(x_i(A_i^{**}), A_i^{**}) \\ -p_i(x_i(A_i'), A_i')] &- \{E_i\{q_i(a^{**},\eta_i)L_i(a^{**})\} \\ -E_i\{q_i(a,\eta_i)L_i(a')\}\}] &\geq 0 \quad \forall A_i' \neq A_i^{**} \end{aligned} \tag{5A-2}$$

where $a^{**} = a(r_1^{**},..., r_n^{**}, W)$, $a' = a(r_1^{**},...,r_i(x_i(A_i')\, A_i'),..., r_n^{**}, W)$, $r_i^{**} = r_i(x_i^{**}, A_i^{**}, v_i)$, and $x_i(A_i^{**})$.

### *An Efficient Liability Rule*

Comparison of (5A-1) with (2A-1) implies that the following liability rule, when applied under strict liability, ensures the marginal conditions for efficiency will be satisfied:

$$L_i(a) = [D(a) + k_i]\left[\frac{g(\cdot)}{q_i(a,\eta_i)h_i(\cdot)}\right] \tag{5A-3}$$

where $k_i$ is a lump sum amount that is yet to be defined. To see that rule (5A-3) leads to the efficient marginal conditions, note that the liability each producer expects to be held responsible for ex ante under rule (5A-3) is

$$\begin{aligned} E_i\{q_i(a,\eta_i)L(a_i)\} &= \iiint q_i(a,\eta_i)\left\{[D(a)+k_i]\left[\frac{g(\cdot)}{q_i(a,\eta_i)h_i(\cdot)}\right]\right\}h_i(\cdot)\,dn\,dw\,d\eta_i \\ &= \iint [D(a)+k_i]g(\cdot)\,dn\,dw \\ &= E\{D(a)+k_i\} \quad \forall i \end{aligned} \tag{5A-4}$$

---

[3] The mathematical foundations for efficient, strict liability rules are developed in this appendix. Unless otherwise stated, the underlying model and assumptions are as developed in Appendix 2A.
[4] Segerson (1995) defines $q$ as a deterministic function of $a$.

Thus, each producer expects to pay an amount equal to total expected damages, plus a constant. The producer's marginal conditions for input use (conditional on technology) will be efficient because taxes of the same form as the left-hand side of (5A-4) have been shown to induce the efficient marginal conditions (Horan et al. 1998a,b; Hansen 1998).

For the special case in which $q_i = 1$ and $g(\nu, W) = h_i(\nu, W, \eta)$, the liability rule defined by (5A-3) becomes uniform. More generally, however, the liability rule in (5A-3) is also a nonlinear function of ambient pollution levels and is site specific. Even though the efficient liability rules are site specific in general, each producer will expect to pay the same variable amount, plus or minus a lump sum amount.

Each producer expects to pay the same variable portion of the liability rule because the liability rule is designed to offset the effects of heterogeneity. In effect, the "correction term" $g(\cdot)/(q_i(a, \eta_i)h_i(\cdot))$ alters the uncertainty that each producer faces about random events and the prospect of being held liable so that each faces the same uncertainty as the resource management agency. Liability must be higher for producers who either believe they will not be found liable (i.e., $q_i\ (a, \eta_i)$ is small) or who feel they do not contribute to ambient pollution levels (i.e., $h_i(\cdot)$ is small) in order to induce them to operate efficiently.

The lump sum part of the liability rule is used to ensure longrun efficiency and to ensure that total payments equal total damages, as is required in a liability framework. For producers operating on extramarginal land, the lump sum component can be set to ensure that these producers expect it to be more profitable to retire extramarginal acreage from production. For producers operating on marginal or inframarginal land, setting the lump sum portion of the expected liability rule as follows ensures that polluters in the region are held liable for total damages

$$k_i = \frac{q_i(a^*, h_i)h_i(\cdot)}{g(\cdot)}[D(a^*)]r_i - D(a^*) \quad \forall i \leq n \qquad \text{(5A-5)}$$

where $\rho_i$ defines the manner in which damages will be distributed among producers ($\sum\rho_i = 1$). Thus, total damages are divided among producers on a site-specific basis, minus a lump sum refund of the variable payment. Because $k_i$ depends on the realization of the term $D\ (a^*)$, $k_i$, is random *ex ante*.

The distribution of payments (i.e., $\rho_i$) can take a variety of forms. For example, suppose that

$$r_i = p_i^* / \sum_{i=1}^{n} p, \text{ where } p = \sum_{i=1}^{n} p_i^*$$

Then, a budget-balancing solution will exist when each producer operating on marginal or inframarginal land expects to earn profits after liability payments

$$p_i^* - E_1\{q_1 D(a^*)\}\frac{p_i^*}{p} \geq 0 \qquad \text{(5A-6)}$$

Condition (5A-6) can be written as

$$[p - E_1\{q_i D\}]p_i^* \geq 0 \qquad \text{(5A-7)}$$

which implies that the ex ante budget-balancing condition is feasible (i.e., no marginal or inframarginal land will be retired due to the expected liability payment) if

$$p \geq E_i\{q_i D(a^*)\} \quad \forall i \leq n \qquad \text{(5A-8)}$$

This condition requires that aggregate pre-liability profits be greater than expected damages when $q_i = 1$, and less than expected damages otherwise. Uniform and other distributions for $\rho_i$ are feasible under more stringent conditions (see e.g., Horan and others, 1998a for a discussion of budget-balancing solutions for ambient-based taxes).

The informational requirements necessary to attain an efficient outcome under strict liability are extreme. The resource management agency must have perfect information on each producer's production, runoff, and joint density functions, and fate and transport. Each producer must also have perfect information on his own as well as other producers' production, runoff, and joint density functions.

# Appendix 5B—Negligence Rules[5]

Under a negligence rule, producers are held liable only if they failed to use the "due standard of care." This standard may be defined either in terms of a damage (or other performance-based) target, $D_0$, or in terms of design (i.e., input use and technology) standards.

First, consider the class of negligence rules based on input use and technology. Define $\Theta_i$ as the set of $A_i$ such that $E\{D(a^*)\} \leq E\{D(a_i^*)\}$, where

$$a_i^* = a(r_1^*,\ldots,r_{i-1}^*, r_i(x_i^*, A_i, n_i), r_{i+1}^*,\ldots,r_n^*, W)\ \forall i$$

Then an optimal negligence rule is

$$N_i(a) = \begin{cases} L_i(a) \ \ \text{if} & \begin{cases} y_i > y_i^*, z_i < z_i^*, A_i \in \Theta_i \ \forall i = 1,\ldots,n \\ y_i > y_i', z_i > z_i', A_i \in \Theta_i' \ \forall i > 0 \end{cases} \\ 0 & \text{otherwise} \end{cases} \tag{5B-1}$$

where $y_i$ is the subset of inputs that increases runoff levels, $z_i$ is the subset of inputs that reduces runoff levels, and $L_i(a)$ is the liability rule defined by (5A-3) and (5A-5).

Efficient entry and exit are ensured by setting input and technology standards $y'_i$, $z'_i$ and $\Theta'_i$ at levels such that profitable operation on acres $i > n$ is not possible without being held liable. Thus, producers who fail to retire extramarginal land will be subject to liability if they attempt to produce at profitable levels. However, because the liability rules, $L_i(a)$, have been designed to ensure efficient entry and exit, production on extramarginal land will not be expected to be profitable. Alternatively, producers operating on marginal or inframarginal land will choose to operate at the efficient level and pay no penalty. If they chose to operate at greater levels of $y_i$, smaller levels of $z_i$, or at technology outside of the set $\Theta_i$, then they would be subject to a significant penalty in the form of the liability rule. Thus, producers expect to be more profitable by operating at the efficient level.

Alternatively, a negligence rule could be based on a damage target, $D_0$. For the damage target, all producers are held liable whenever $D(a) > D_0$. Otherwise, no producer is held liable. Consider the following negligence rule

$$N_i(a, D_0) = \begin{cases} \{L_i(a) & \text{if} \quad D(a) > D \\ 0 & \text{otherwise} \end{cases} \tag{5B-2}$$

where $L_i(a)$ is again the liability rule defined by (5A-3). The $k_i$ terms (in $L_i(a)$, see 5A-3), however, are not necessarily defined as in (5A-6). In the strict liability case, the $k_i$ terms are constructed to ensure efficient entry and exit and that the aggregate liability equals total damages. However, such a construction will not necessarily be effective in limiting entry when a negligence rule is imposed because it is not possible to target specific polluters and specific production practices as it is with the negligence rule (5B-1). Instead, polluters may all avoid liability by producing at suboptimal levels (Miceli and Segerson, 1991), and production may be profitable on more than the efficient number of acres without the threat of a liability penalty.

[5] The mathematical foundations for efficient negligence rules are developed in this appendix. Unless otherwise stated, the underlying model and assumptions are as developed in Appendix 2-A.

## Chapter 6

# Education

*Education is used to provide producers with information on how to farm more efficiently with current technologies or new technologies that generate less pollution and are more profitable. While such "win-win" solutions to water quality problems are attractive, education cannot be considered a strong tool for water quality protection. Its success depends on alternative practices being more profitable than conventional practices, or on the notion that producers value cleaner water enough to accept potentially lower profits. Evidence suggests, however, that net returns are the chief concern of producers when they adopt alternative management practices. In this chapter, we review the economic framework behind education, and review the empirical evidence for the potential role of education in a pollution control policy.*

## Introduction and Overview

Education plays a significant role in many State and Federal nonpoint-source water quality programs, most recently in the Clean Water Action Plan (EPA-USDA, 1998; Nowak and others, 1997). Educational programs are designed to provide agricultural producers with better knowledge about production relationships for current technologies (so that inputs can be used more efficiently) and/or about alternative technologies that may be more profitable and pollution-abating. In addition, producers may be shown how they contribute to nonpoint pollution and how this may affect themselves and others. Methods for conveying information include demonstration projects, technical assistance, newsletters, seminars, and field days.

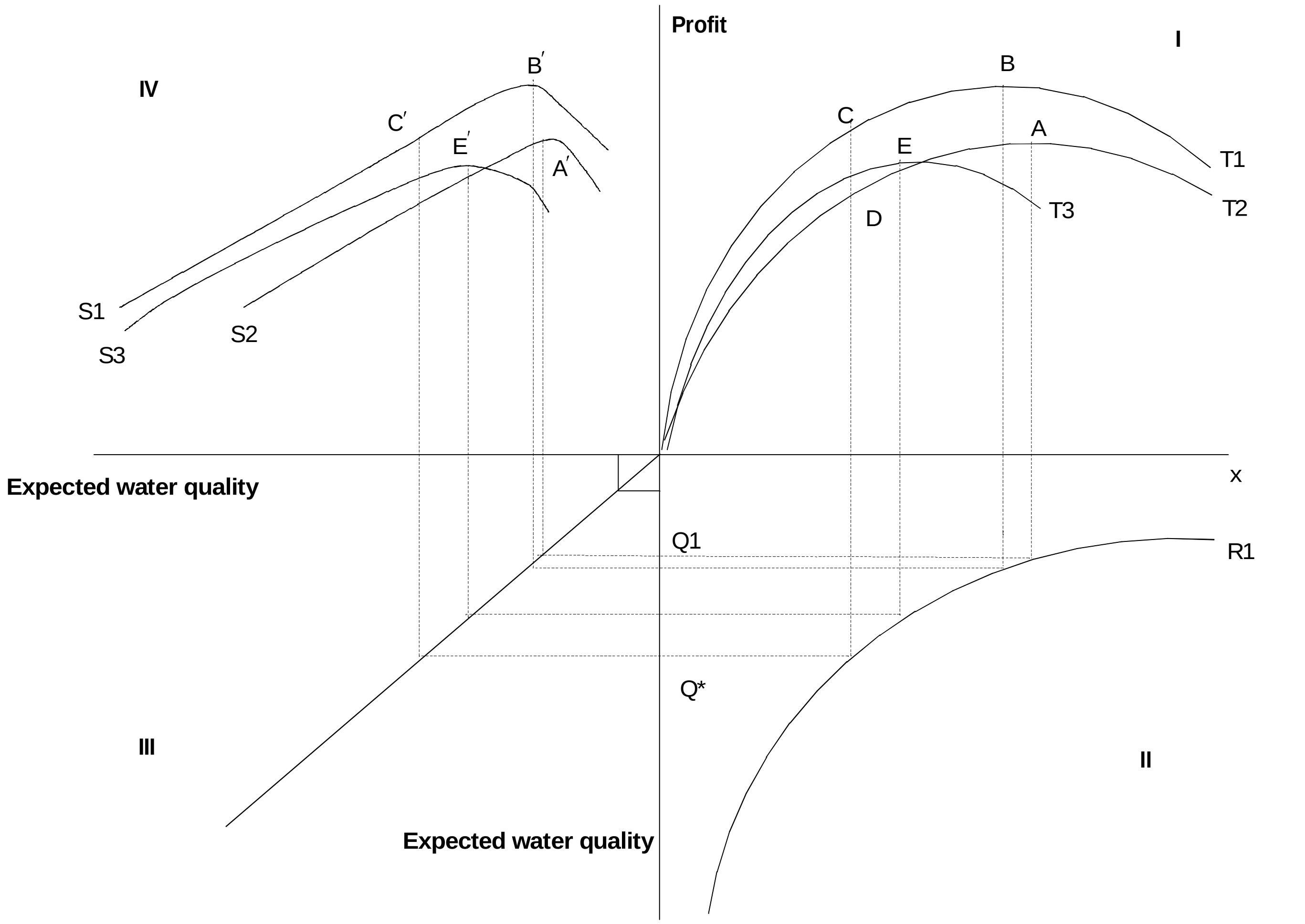
Education is popular as a nonpoint strategy for a number of reasons. It is less costly to implement than a cost-share program, and the infrastructure for carrying out such a program is largely in place (county extension, Natural Resources Conservation Service field offices, land grant universities). Education has been effective in getting producers to adopt certain environmentally friendly practices (Gould, Saupe, and Klemme, 1989; Bosch, Cook, and Fuglie, 1995; Knox, Jackson, and Nevers, 1995). Specifically, educational assistance is often seen as a means of achieving "win-win" solutions to water quality problems, whereby information encourages producers to operate in ways that improve both net returns and water quality (EPA-USDA, 1998; EPA, 1998a). Some practices that have been shown to achieve both aims include conservation tillage, nutrient management, irrigation water management, and integrated pest management (Bull and Sandretto, 1995; Ervin, 1995; Conant, Duffy, and Holub, 1993; Fox and others, 1991).

This chapter begins with education's role in changing producers' expectations about the performance of current technologies. Next, we show how education works under different levels of stewardship or altruism on the part of the producer, and with different levels of private benefits generated by water quality-protecting practices. We then present evidence that education programs have had a limited impact on changing producer behavior when water quality practices are promoted.

## Assessing Education as a Water Quality Protection Tool

Figure 6.1 depicts the relationships between production and expected water quality for a single farm, (which may be one of many contributors to nonpoint pollution in a watershed), for the simplified case in which a single input leads to water quality impairment. The relationship between input use and the producer's net returns (i.e., the restricted profit function) is illustrated in quadrant I. Without loss of generality, the profit (y) axis could be thought of as the expected utility of profits for risk-averse producers when there is production uncertainty. Tradeoffs would then be made between expected utility and expected water quality. The relationship between input use on the farm and expected

Figure 6.1
**Producer production decisions, without altruism**




water quality, taking the actions of all other nonpoint polluters as given, is represented in quadrant II. Finally, the relationship between expected water quality and net returns—or how producers account for water quality in their production decisions—is quadrant IV. A utility indifference map showing the rates at which a producer is willing to trade net returns for increased water quality can be constructed. The point along the water quality–net returns frontier where a producer will operate is at the point of tangency with an indifference curve, or where the marginal rate of substitution (MRS) between net returns and water quality is equal to the slope of the net returns–water quality frontier. At this point, the producer's utility is maximized.

Producers commonly face varying degrees of uncertainty in many aspects of production. For a given production technology, uncertainty about the production frontier (i.e., how to attain the greatest yield or profit levels for a given combination of inputs) may lead producers to use inputs inefficiently. This situation is represented by curve T2, which reflects the production technology the producer is currently using (i.e., the set of tillage, pest control, nutrient management, and conservation practices used to grow a particular crop or set of crops), and the skill with which he is using it. Producers may also have limited knowledge about alternative production technologies and their economic and environmental characteristics, as well as about how their production decisions affect water quality.

The resource management agency's (RMA's) expectations about the relationship between net returns and input x are defined by T1. The RMA's beliefs about the relationship between input use and potential profits are assumed to be more accurate than the producer's due to publicly supported research on how x can be used more efficiently than under the producer's current technology set. The RMA may also have better information about alternative technologies (which could also be represented by T1) and about the relationship between input use and water quality (curve R1).

Suppose the Pareto-efficient level of expected water quality is at Q* (with production occurring at point C on curve T1), but that existing expected water quality levels are well below this. Such inefficiencies arise when (1) producers do not consider the economic impacts of their production decisions on water quality and/or (2) producers face uncertainty or have a limited understanding of the production and environmental impacts of their management choices. The purpose of educational programs is to reduce producers' uncertainty and to improve their knowledge about production and environmental relationships (both for current and alternative technologies). Proponents of such programs believe expected water quality will be improved if the information provided encourages producers to (1) consider the environmental impacts of their choices and/or (2) simultaneously improve expected water quality and profitability by using existing technologies more efficiently or by adopting alternative, more environmentally friendly technologies (Nowak and others, 1997).

## Education's Appeal to Profit, Altruism, Efficiency

Below, we discuss the ability of educational programs to provide incentives for improving expected water quality. For simplicity, we ignore shortrun influences such as risk and learning. Instead, we take a longrun view and assume that a practice will eventually be adopted if education can convince producers that it will make them better off (increase expected utility). We note, however, that uncertainty and other factors could slow or prevent the adoption of practices that might, in the long run, increase producers' net returns and improve water quality (see chapter 3). Such factors represent additional limitations that educational programs would have to overcome.

### *No private benefits from water quality improvement and no altruistic/stewardship motives*

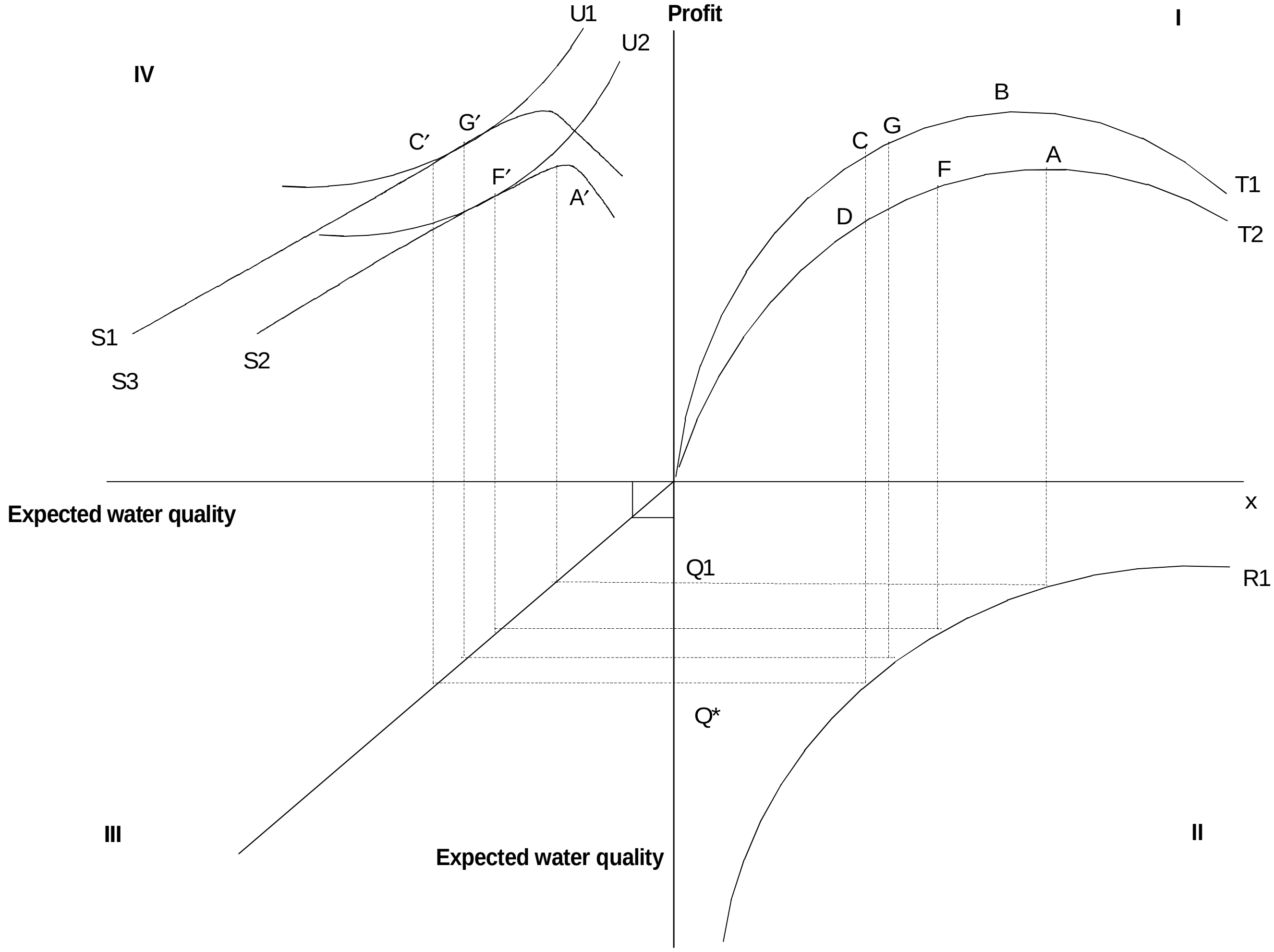
Suppose a profit-maximizing producer who, due to production uncertainty, produces inefficiently along T2 at point A in figure 1. The producer is assumed to receive no private benefits from environmental improvement (i.e., chemical use does not affect the quality of the producer's water supply or of recreation areas the producer visits) and to have no altruistic or stewardship motives (i.e., the producer does not include social damages in his decision set). In this setting and in the absence of any outside programs or intervention, the producer would not voluntarily move to point D so that the RMA's goals are achieved, since net returns would be reduced without any compensating private benefits. In quadrant IV, the MRS between net returns and water quality is 0 (horizontal line), and producers operate at point A′ (where the slope of S2 is 0).

How might education encourage more efficient resource use and improve expected water quality in this situation? It would be pointless for the RMA to educate the producer about the relationships between production and water quality since the producer has no altruistic or stewardship motives. However, by educating the producer about the frontier T1, where profits are higher for each level of input use, the RMA could encourage the producer to use existing management practices more efficiently or to adopt alternative practices so that he/she operates along T1.

Once on T1, the producer could operate at the Pareto-efficient point C to meet the expected water quality goal and at the same time increase net returns relative to operation at point A on T2 (although there may be values of C for which net returns might be reduced). Such an outcome appears to be a "win-win" solution for the farmer. However, even though the producer is producing along a more socially efficient production frontier, his/her goals of production will still generally differ from society's. As long as producers consider only profitability, the producer will operate at point B (note that point C is necessarily to the left of B). The expected water quality levels that correspond to B are an improvement over A, but are still less than efficient. Thus, educational assistance alone is not enough to ensure that the water quality goal is met.

Providing education about production practices might even reduce expected water quality. Suppose the producer originally produced according to T3, so that profits were maximized at E. After receiving educational assistance, the producer would have an incentive to produce at point B on T1. Net returns increase in this case, but so does the use of input x. The result is that expected water quality is worse than it was before education was provided. This result is more than just a curiosity. There is evidence that some IPM practices have actually

Figure 6.2
**Producer production decisions, with altruism**




increased the amounts of pesticides producers use (Fernandez-Cornejo, Jans, and Smith, 1998).

### *Altruistic/stewardship motives*

Producers may have altruistic or stewardship motives when it comes to the effects of their production decisions on others and on the environment. They may be willing to sacrifice some net returns in order to protect water quality. If so, then education that encourages producers to broadly consider the consequences of polluting practices on water quality and on water users may be somewhat effective. Research has demonstrated that producers are often well informed of many environmental problems, and that most U.S. producers hold very favorable attitudes toward the environment and perceive themselves as stewards of the land (Camboni and Napier, 1994). Educational programs could take advantage of altruism or stewardship by informing producers about local environmental conditions and about how a change in management practices could improve local water quality. This would be accomplished by providing producers with information about T1, and also about the relationship between their production practices and water quality, R1.

Suppose an altruistic producer does not believe he/she is contributing to water quality problems and is not aware of T1 (fig. 2). Production will initially take place along T2 at A (or at A′ in quadrant IV). Since the producer is unaware of R1, the producer's MRS between net returns and water quality is 0. Suppose that the producer is informed of how the use of x is affecting water quality (becomes aware of the relationships expressed by R1 and S2). Where the producer now operates will be determined by his/her willingness to give up some net returns to protect water quality, expressed by the indifference curves in quadrant IV. Production on T2 will now occur to the left of A, at F (F′ in quadrant IV), where indifference curve U2 is tangent to S2. In the example, water quality is improved and utility increased (point A′ lies below U2). This is a win-win situation for the producer in terms of utility, even though net returns are reduced relative to A.

Suppose now the producer is educated about T1. The altruistic producer will have an incentive to make production decisions based on the tradeoffs defined by S1 and U1, operating now at point G. In this example, both water quality and net returns are higher than for points A and F, a win-win situation. However, this need not be the case. The ultimate impacts to water quality will generally depend on the nature of T1 and R1 relative to T2, and on the MRS between net returns and water quality. If expected water quality does improve as a result of education, the degree of improvement relative to the RMA's goal of Q* depends on how strongly the producer values environmental quality. Efficiency is obtained only for the special case in which each producer makes production decisions while fully internalizing his/her marginal contribution to expected environmental damages.

Experience with education programs indicates that altruism or concern over the local environment plays only a very small role in producers' decisions to adopt alternative management practices. Agricultural markets are competitive, and at a time when commodity program payments to producers are being reduced and trade is being liberalized, market pressures make it unlikely that the average producer will adopt costly or risky pollution control measures for altruistic reasons alone, especially when the primary beneficiaries are downstream (Bohm and Russell, 1985; Abler and Shortle, 1991; Nowak, 1987; Napier and Camboni, 1993). A survey of Pennsylvania field crop producers found that private profitability was the motivating force in adopting environmental practices, although altruistism was also a determinant (Weaver, 1996). Camboni and Napier (1994) found that education was not effective in promoting adoption of practices that were less profitable than current practices.

USDA's Water Quality Demonstration Projects—now discontinued—provided educational assistance to producers in 16 areas where agriculture was known to be affecting water quality (Nowak and others, 1997). A study of producer adoption of improved farming practices for protecting water quality was conducted using a sample of these projects. It compared adoption rates of similar management practices in the Demonstration Project areas and in control areas where education was not provided, and found little difference in awareness, familiarity, and adoption. In fact, only 1 case out of 20 showed a significantly greater adoption rate in the Demonstration Projects than in the comparison sites during 1992-94 (Nowak and others, 1997). It is possible that information spillovers from the Demonstration Projects influenced the control sites, but it is just as possible that producers are generally looking for management practices that increase net returns and that education alone was inadequate for accelerating the adoption of practices that protect water quality.

In another example, California's Fertilizer Research and Education Program, a voluntary nitrate management program, has not had much success in altering fertilizer management practices, despite well-publicized groundwater quality problems (Franco, Schad, and Cady, 1994). More public supply wells in California have been closed for nitrate violations than for any other contaminant. Four years of education efforts have not fundamentally changed fertilizer management practices. To date, appeals to stewardship have not overcome concerns over maintaining high yields.

Altruism can motivate change only if producers believe there is a problem that needs to be addressed and that their actions make a difference (Napier and Brown, 1993; Padgitt, 1989). Surveys consistently find that producers generally do not perceive that their activities affect the local environment, even when local water quality problems are known to exist (Lichtenberg and Lessley, 1992; Nowak and others, 1997; Pease and Bosch, 1994; Hoban and Wimberly, 1992). Producers' perceptions about their impacts on water quality did not significantly change over the course of USDA's Demonstration Projects, even though the projects were located in areas with known water quality problems (Nowak and others, 1997). This indicates either a lack of effort to educate producers on their role in protecting local water quality or the difficulty of convincing producers of their role in solving the problem.

Convincing producers of their contribution to a nonpoint-source pollution problem is inherently difficult. Nonpoint-source pollution from a farm cannot be observed, and its impacts on water quality are the result of a complex process and are often felt downstream from the source. If there are many other producers in the watershed, a single producer may justifiably believe that his/her contribution to total pollution loads is very small. This means that producers will have to take as a matter of faith the RMA's description of the relationship between production and water quality, R1. Even if a producer does take appropriate actions to improve water quality, he/she generally will not be able to observe whether these changes in management actually improved water quality. Once again, the producer will have to take as a matter of faith any information the RMA may provide about the impacts of his/her efforts to improve water quality.

### *Practices generate private benefits from water quality improvement*

There are cases in which a producer's practices may affect the farm's drinking water supply and his/her family's health, or in which water quality influences onfarm productivity. A producer in such a situation may be willing to forgo some profit for an increase in expected water quality if the expected onfarm environmental impacts are sufficiently large (expected utility from profits and water quality increase). Therefore, an educational program that addresses these onfarm environmental impacts may motivate the producer to change production practices to improve expected water quality. The analysis is similar to that for altruistic farmers, except the impacts on water quality are felt closer to home and it is probably easier for the RMA to establish the consequences of polluted water. This is illustrated in figure 2 as the producer moves from A toward D on T2 after being informed of the potential onfarm impacts. The actual point of production relative to D depends on the perceived significance of the risk and the value placed on that risk, reflected by the indifference curves. If the producer is also provided with information on T1, he/she will have incentive to operate along T1 somewhere to the left of B. Both producer utility and water quality increase as a result of education.

If onfarm impacts were the only possible water quality problems from farming, then consideration of these impacts in production decisions would result in an efficient allocation of resources (there is no externality). However, if onfarm impacts are being used by the RMA as a proxy for other offsite impacts, then inefficiencies would still exist in the allocation of production resources. An analysis of the impact of user safety concerns over herbicides used on corn and soybeans in four States found that herbicide toxicity did not have a sizable impact on herbicide use decisions (Beach and Carlson, 1993). The herbicides used were generally not very toxic to humans, and productivity effects dominated herbicide use decisions. Decisions based on protecting human health were inadequate for protecting environmental quality.

Producers have been shown to respond to education programs when their own water supply is at stake (Napier and Brown, 1993). This is demonstrated by the Farm*A*Syst program. This program, developed by the Wisconsin Cooperative Extension Service and supported by USDA, teaches producers to assess impacts of farming operations around the farmstead (Knox, Jackson, and Nevers, 1995). Educating producers raises their self-interest for altering certain practices, primarily around private wells. Producer education has succeeded in getting individuals to take cost-effective actions to remediate problems from leaking fuel storage tanks, pesticide spills, and drinking water wells contaminated by runoff from confined animals. Studies from Minnesota, Wisconsin, and Louisiana show producers to be receptivite to Farm*A*Syst and voluntarily willing to take action to reduce high risks by changing management practices and facility design (Knox, Jackson, and Nevers, 1995; Anderson, Bergsrud, and Ahles, 1995; Moreau and Strasma, 1995). The key to the program's apparent success is the ability to identify the source of a threat to the producer, his family, and his employees.

## Education and Industry Structure

Educational programs do not influence decisions about entry and exit into the industry. Acreage that would be classified as extramarginal in the efficient or cost-effective solution may still remain in production if educational assistance is the only form of government intervention. It is unlikely that any producer would voluntarily retire land from production if provided information on alternative practices or how his/her operation may be affecting water quality.

Current USDA education programs unwittingly may be disproportionately helping larger farms. Small producers have been found to be less likely to adopt new practices than large producers (Lichtenberg, Strand, and Lessley, 1993; Ervin and Ervin, 1982; Gould, Saupe, and Klemme, 1989; Norris and Batie, 1987). A study of producers around the Chesapeake Bay found that cost sharing and subsidized technical assistance were used much more by larger farms than smaller ones to adopt nutrient management, animal waste management, and soil erosion control (Lichtenberg, Strand, and Lessley, 1993). Smaller farms may face tighter credit constraints and be more risk averse. Education efforts may also be directed more at larger farms, based on the assumption that changing practices on these farms would generate the greatest environmental benefits. If new practices enhance net returns, then larger farms benefiting from education efforts may be putting smaller farms at greater economic disadvantage relative to larger farms. This may conflict with a societal goal of protecting small, resource-limited producers.

## Summary

Water quality policies based on education are currently popular because education is a benign form of intervention (i.e., producers are not forced to change their management), it is relatively inexpensive to administer, and it may teach producers how to achieve higher returns. From a practical standpoint, the institutional structure necessary to implement this approach—USDA, State conservation agencies, and land-grant institutions—is already in place (Easter, 1993). If education succeeds in raising a producer's awareness about a local environmental problem, and the producer places a value on protecting environmental health, the effect on producer willingness to adopt alternative practices can be significant.

However, education has some important shortcomings in achieving the water quality levels demanded by the public, even when ignoring the short-term constraints to adoption. Educational programs will improve water quality only if the information provided to producers encourages them to take actions that lead to water quality improvements. Such incentives exist when (1) the actions that improve water quality also increase profitability, (2) producers have strong altruistic or stewardship motives, and/or (3) the onfarm costs of water quality impairments are shown to be sufficiently large. However, none of these three conditions guarantees an expected improvement in water quality. In general, the outcome of educational programs depends on how actual profitability–water quality frontiers compare with the producer's initial understanding of these frontiers. Moreover, in the absence of altruistic or stewardship motives, alternative practices that simultaneously increase expected net returns over the long term and improve water quality are very few.

Many education programs may not devote enough effort to convincing producers of their role in water quality protection. Failure to do so limits the extent to which stewardship influences producer decisions. The influence of stewardship is also probably limited by the longrun financial viability of the farming operation, including current and anticipated risks. If the socially efficient outcome can be achieved only through significant reductions in producer net returns, then education will probably not be effective in achieving the desired water quality goal, even if producers understand the relationship between production practices and water quality.

## Chapter 7

# Research and Development

*Research and development are important tools in reducing agricultural nonpoint-source pollution because they provide producers and society with more efficient ways of meeting environmental goals. However, producers and private firms will likely underinvest in research and development on improving water quality. Public involvement is therefore necessary either to carry out this research or to provide producers and the private sector with incentives (economic incentives or regulations) that result in more efficient research investments. Finally, R&D cannot independently provide a solution to water quality problems. Instead, it is a valuable component of other approaches.*

## Introduction and Overview

Extensive public and private resources are devoted each year in the United States to agricultural research and development. Research and development can provide producers with new or improved inputs, technologies, and management techniques that can address concerns such as productivity, net income, and environmental quality. In this chapter, we discuss the role of research in reducing water pollution generated by farming and the factors that generate demand for innovation. We show that incentives for private research are inadequate because many benefits of research are not captured by private markets. In other words, there are social benefits from research that do not result in returns to investors. Consequently, research will be underfunded relative to levels that would occur if investors were to consider these additional social benefits. Government can provide incentives for private research by establishing a system of intellectual property rights, and fund research that produces goods that are public in nature.

This chapter begins by discussing the types of innovations that can reduce water pollution from agriculture. Next, we show why appropriate incentives do not exist for investment in research leading to innovations to improve water quality when there is no government intervention to correct externalities. We then show how policies based on standards and economic incentives create incentives for private research. Finally, government's role in the research and development process is discussed, along with a description of how public support has influenced research and development programs in the United States.

## Innovations That Improve Water Quality

Innovations having positive water quality impacts can broadly be classified as (1) augmenting factors, (2) reducing pollution, or (3) introducing entirely new inputs and technologies (see table 7-1), although an innovation may exhibit aspects of each.

**Factor-augmenting innovations** allow the same quantity of output to be produced with less of the augmented factors (i.e., inputs). Examples related to nonpoint pollution include more effective pesticides and fertilizers, new seed varieties that are higher yielding or require fewer inputs, and enhanced irrigation efficiencies. Factor-augmenting innovations may result in reduced use of polluting inputs and, consequently, reduced runoff and ambient pollution levels. This may not always be the case, however. The use of polluting inputs may increase due to input substitutions and changes in the scale of production. In a simulation of U.S. corn production, Abler and Shortle (1995) found that capital-augmenting innovations would increase fertilizer and pesticide use. They also found that pesticide use would be increased by land- and seed-augmenting innovations and decreased by pesticide-augmenting technologies.[1]

[1] Abler and Shortle's results were driven largely by the high elasticity of demand for corn.

**Table 7-1—Types of innovations**

| Innovation type | Example |
|---|---|
| Factor-augmenting technology | Soil-nitrogen testing |
| | Integrated Pest Management |
| | Split nitrogen application |
| | Nitrogen breakdown inhibitors |
| | Subsurface micro-irrigation |
| | Conservation tillage |
| Pollution-reducing (runoff abatement technology) | Buffer strips |
| | Sediment basins |
| | Microbial phytase (feed additive) |
| Entirely new inputs | New pesticides and other chemicals |

**Pollution-reducing innovations** have no impact on crop production relationships, but they do reduce runoff (and hence pollution) for any level of input use. This type of innovation is essentially an improvement in runoff abatement technology. For example, a pollution-reducing innovation may increase buffer strip effectiveness in filtering out nutrients before they reach a water body.

Advances in science may result in the **introduction of entirely new inputs** to agricultural production. For example, research on extracting atmospheric nitrogen for manufacturing explosives resulted in the introduction of inorganic nitrogen fertilizers to agriculture. Other examples related to nonpoint pollution include satellite and computer technologies for increasing precision application of chemicals and the development and introduction of new crops. Such innovations will likely result in producers' using new combinations of existing inputs and changing the scale of production (or possibly shifting to alternative commodities). Economically attractive innovations that allow producers to completely substitute polluting inputs with alternative technologies will improve environmental quality.

## Private Incentives for Water Quality R&D

Research and development (R&D) is a process by which investment in scientific study leads to future technological innovations. Research programs may proceed along a variety of paths. For example, crop pest control may be improved by genetically enhancing the pest-resistance qualities of a particular crop, by enhancing current or discovering new pesticides, or by developing alternative cropping systems. Unfortunately, innovations are uncertain in terms of timing (if they occur at all), required investment costs, and importance. In the example above, the importance of an innovation in genetic research might refer to the amount of increased pest resistance relative to that of existing crop varieties. Years of effort may result in only a marginal improvement (if any) over existing crop varieties.

Even in the absence of externalities, R&D programs will be underfunded without government intervention to ensure that innovators receive the economic benefits from the sale of the innovation. Underfunding occurs because the results of research often have the characteristics of a public good. Specifically, once an innovation occurs, it is not always possible to exclude others from acquiring the knowledge to use the innovation. Without a legal claim to this knowledge (e.g., a patent or copyright), only a share of the total economic benefits can be captured by private research organizations that develop innovations (Fuglie and others, 1996). A potential problem with intellectual property rights is that they convey monopoly power to the developers of new innovations (Fuglie and others, 1996; Moschini and Lapan, 1997). Under monopoly conditions, use of the innovation will generally be less and the price higher than if it were provided under perfect competition. The intellectual property right may reduce the social value of the innovation, but it is better than not having the innovation at all (Fuglie and others, 1996).

### Market-Based Incentives and Externalities

Given that mechanisms are in place to protect innovators, private incentives for investment in R&D exist. Economic theory and empirical evidence show research organizations have incentives to invest in agricultural research devoted to factor augmentation or new innovations that shift production from relatively scarce (or costly) inputs toward relatively abundant (or cheaper) inputs (Hayami and Ruttan, 1985; Ruttan and Hayami, 1989; Antle and McGuckin, 1993). Continuing with the pest-resistance example, suppose current pest control methods rely heavily on pesticide use. A relative increase in pesticide prices creates an incentive to invest in any of the aforementioned research paths (i.e., genetically enhancing crops, altering cropping practices, etc.) that promise to reduce pesticide costs.

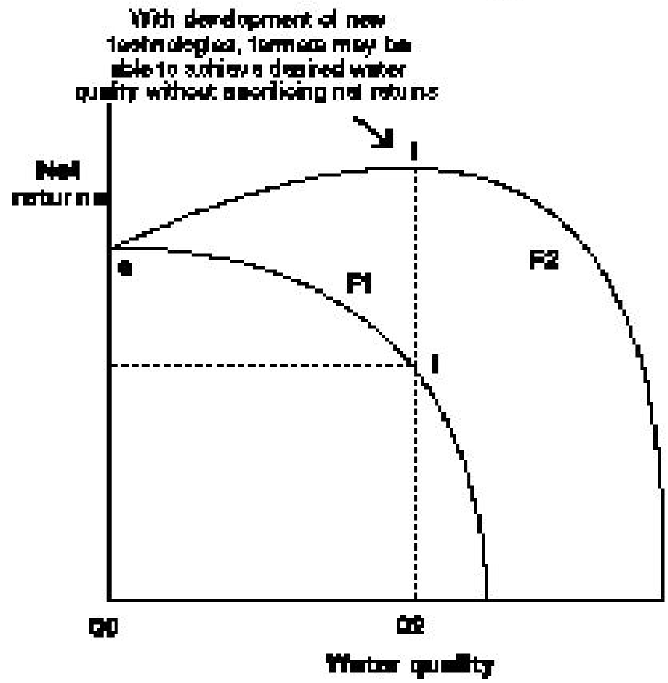
Moreover, economic incentives (created by market or institutional forces) are important determinants of the expected private return to investment for each potential research path. Consequently, these incentives also play an important role in the allocation of investments for each path.[2] For example, the expected marginal return to pesticide research may be small relative to that of genetic engineering research if chemical restrictions are expected to become more stringent relative to regulations on genetic-engineered products. Increased regulation of pesticides to make them safer to farmworkers and to the environment may have reduced the introduction of new materials (Ollinger and Fernandez-Cornejo, 1998).

Inputs that create (inhibit) nonpoint pollution are underpriced (overpriced) without government intervention because private markets do not reflect the social costs of input use. Private research organizations therefore do not have the economic incentives to invest efficiently in R&D programs that may lead to innovations in improving water quality.[3] For example, heavy use of nutrients in agriculture is widespread because nutrients have historically been relatively inexpensive and government regulation of the externalities caused by their use has been minimal or nonexistent. Consequently, incentives to develop new crop varieties that require fewer nutrients are not strong. Although nutrients have been seen as inexpensive in private markets, the social costs of nutrient use have been higher because they contribute to nonpoint pollution. R&D may have evolved along another path had nutrients been priced more appropriately.

## Producer Incentive To Adopt Innovations

The incentives for private R&D on pollution-reducing innovations are virtually nonexistent without government intervention, even with intellectual property

Figure 7.1
**Tradeoff between net returns and water quality with introduction of new technology**



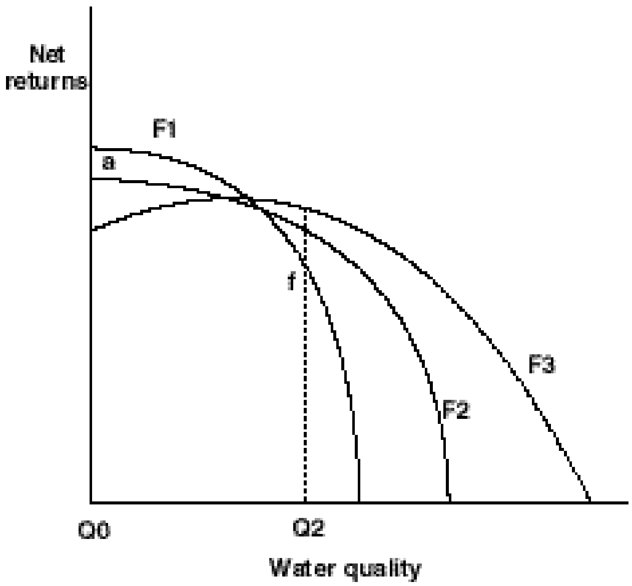

rights. Pollution-reducing innovations are not likely to generate private benefits to producers because they have no positive impacts on profitability.

If incentives are inadequate for private research on innovations that improve water quality, the public sector can fund research on such innovations. Even if such innovations occur, however, there is no assurance that producers will adopt them. In this case, the innovation would not be truly successful. A producer's adoption decision depends greatly on profitability. In a competitive market without government intervention, producers who consider water quality impacts may lose a competitive edge because of the inherent tradeoff between profitability and water quality. Figure 7.1 illustrates that a water quality innovation would need to change the shape of the water quality-net return frontier from F1 to F2 (so that a producer maximizes profit at point i as opposed to point a) in order for adoption to be profitable. In this case, both water quality and profitability are improved. However, profitability must still be weighed against the cost of adoption and the profitability of existing technologies and other innovations.

[2] Assuming investors are risk-neutral and profit-maximizing, investment will occur where marginal expected returns are equated across each path. Factors that influence expectations about returns include the probability of a successful innovation, the expected importance of the innovation, and other relevant economic and institutional factors (such as the current or expected policy environment).
[3] The social effectiveness of research can be measured by the social rate of return on research, defined as the social benefit/cost ratio of research (Fuglie et al., 1996). Research on environment-enhancing technologies compares poorly with other research opportunities when environmental benefits either are not considered or are undervalued.

Figure 7.2

**Tradeoff between net returns and water quality with introduction of new technology and an expected water quality constraint**




## Government Intervention Changes Incentives for Water Quality R&D

Even with the appropriate signals, the private sector will underinvest in environmental research due to its nature as a public good. Private research will focus on innovations that it can control, such as new chemicals, nutrients, machinery, and plant varieties. Research on management-oriented innovations, such as timing nutrient applications, rotations, and tillage practices, will most likely be carried out in the public sector. Furthermore, public sector R&D aimed at developing cheap and effective water-quality-monitoring techniques and devices could remove barriers now preventing the efficient use of standards. An effective R&D policy must remain responsive to price and regulatory signals provided by the economy and society (Fuglie and others, 1996).

Effective intervention requires that investment incentives be altered to reflect the costs that nonpoint pollution imposes on society. Investment incentives can be altered by policies that either assign prices to externalities or increase the relative price of pollution-causing inputs or technologies (see chapters 3 and 4). Regulations and economic incentives are one way of increasing the price of polluting inputs relative to nonpolluting inputs. The increased relative price of polluting inputs causes producers to seek alternative practices that require less of these inputs. For example, producers would benefit from innovations that shift the frontier in figure 7.2 from F1 to F2 or F3 when a standard requires that production results in an expected water quality level of *Q2*.[4] Regulations and economic incentives therefore provide producers and their input suppliers with incentives to invest in research that considers more effective ways of meeting environmental objectives and to adopt resulting innovations.

There is a qualitative difference in the ability of economic incentives and standards to provide incentives for research. Economic performance or design-based incentives provide a "reward" for continued reduction in polluting activities in the form of reduced tax burden or increased subsidy. Standards, on the other hand, do not provide incentives to improve water quality beyond the level defined by the performance standard or the design standard. There is no "reward" for providing an extra measure of control. For example, if a standard is set for a polluting input and a producer is already meeting the standard, there is no demand from the producer for innovations that result in less of the input being used. If instead a tax is placed on the input, an incentive is created for innovations that result in less of the taxed input being used, regardless of a particular water quality goal.

Applying incentives or regulatory policies to different bases will provide different incentives for investment in R&D. Bases that are closer to the externality (i.e., performance bases, expected runoff) are generally more effective in providing the appropriate incentives for investment in each of the three types of innovations. Input- and technology-based instruments are somewhat effective in promoting investment in factor-augmenting innovations and the development of new inputs, depending on the impact the innovation will have on profitability relative to water quality (that is, incentives will be smaller for innovations that lead to improved water quality but do not enhance production). However, input- and technology-based instruments that are related only to production do not induce producers to consider water quality impacts of innovations that are not related to production and do not have positive impacts on profitability. Therefore, input- and technology-based instruments may produce only small incentives for investment in pollution-reducing innova-

[4] Neither of these technologies would be attractive if there were no constraints on water quality.

**Table 7-2—Incentives from different instrument bases for investment in water-quality-improving innovations**

| Instrument base | Factor-augmenting | Pollution-reducing | New inputs |
|---|---|---|---|
| Performance-based | Good | Good | Good |
| Design-based | | | |
| Expected runoff | Good | Good | Good |
| Input- and technology-based | Fair-Good | Poor | Fair-Good for inputs that enhance production; poor for inputs that do not enhance production (i.e., pollution-control inputs) |

Note: These rankings are subjective, based only on theoretical properties as opposed to empirical evidence. A more reliable table would be based on empirical results that compare each type of policy according to a consistent modeling framework that is representative of the nonpoint problem.

tions or the development of new inputs that affect only water quality (and not productivity).

Effective government intervention also must provide producers with the appropriate incentives to *adopt* innovations that provide cost-effective pollution control. As shown in chapter 2, producers would have an incentive to adopt the most socially efficient innovations if all externalities were priced at their efficient levels. Applying incentives or regulatory policies to different bases will provide different incentives for the adoption of innovations. The adoption incentives provided by each base (table 7-3) are almost identical to those provided for R&D investment (table 7-2). Bases that are closer to the externality are generally more effective in providing the appropriate incentives for the adoption of each innovation type, including pollution-reducing innovations. Input- and technology-based instruments are somewhat effective in promoting adoption of factor-augmenting innovations and the development of new inputs, depending on the impact the innovation will have on profitability relative to water quality. In addition, input- and technology-based subsidies and standards are likely to be effective in inducing producers to adopt pollution-reducing innovations that are not related to production because these instruments make it profitable (or necessary) for producers to consider these impacts.

The second-best incentive or regulatory policies that are most likely to be implemented (due to the information, administration, and implementation costs associated with efficient policies) will not necessarily provide producers with incentives to adopt cost-effective water quality innovations as they become available. When input-based standards or economic incentives are used, the resource management agency needs to adjust the standards or incentives on all inputs or technology to reflect the new innovations. Not doing so will result in a level of pollution control that is not cost effective.

## Has Research Helped?

Public and private research has had a few successes in developing complementary technologies that enable producers to both achieve water quality improvements and increase net returns. For example, some Integrated Pest Management (IPM) categories use enhanced information and multiple pest control strategies (chemical, biological, and cultural) to manage pest populations in an economically efficient and ecologically sound manner. A review of 61 farm-level economic evaluations concluded that IPM was generally profitable (U.S. Congress, OTA, 1995). This finding is supported by the fact that more than half the fruit, nut, corn, soybean, and fall potato acreages were using an IPM approach during 1991-1993 (Vandeman and others, 1994).

Conservation tillage is a family of tillage practices that leave at least 30-percent of the planted soil surface covered by crop residue to reduce soil erosion by water and polluted runoff (U.S. Congress, 1995). Conservation tillage has been shown to be profitable for a number of crops in many areas (Fox and others,

**Table 7-3—Incentives from different instrument bases for adoption of water quality-improving-innovations**

| Instrument base | Factor-augmenting | Pollution-reducing | New inputs |
|---|---|---|---|
| Performance-based | Good | Good | Good |
| Design-based | | | |
| Expected runoff | Good | Good | Good |
| Input- and technology-based | Good with subsidies or standards. Otherwise, fair-good | Good with subsidies or standards. Otherwise, poor | Good with subsidies or standards. Otherwise, fair-good for inputs that enhance production; poor for inputs that do not enhance production (i.e., pollution control inputs) |

Note: These rankings are subjective, based only on theoretical properties as opposed to empirical evidence. A more reliable table would be based on empirical results that compare each type of policy according to a consistent modeling framework that is representative of the nonpoint problem.

1991). As a result, its use has steadily grown in recent years (USDA, ERS, 1997).

Another technological innovation that improves water quality is improved soil nitrogen testing. This enables more accurate nitrogen applications, resulting in fewer over-applications and consequently less runoff and subsurface leaching. This technology is most appropriate where there has been a history of manure applications (Fuglie and Bosch, 1995; Musser and others, 1995). A related technology, subsurface micro-irrigation, reduces water use and can place nutrients more precisely in the root zone compared with center-pivot irrigation. It is more profitable than conventional center pivot irrigation on small fields, but not on large fields (Bosch, Powell, and Wright, 1992) and also results in reduced runoff and leaching.[5]

Other new technologies that may result in improved water quality are not yet profitable and will require a subsidy or regulation to become widely used. For example, microbial phytase as a feed additive can reduce phosphorus in swine and poultry excretions by 50 percent or more (Simons and others, 1990; Coelho and Kornegay, 1996). Similarly, USDA's Water Quality Program discovered several new or improved methods of applying pesticides and fertilizers for corn-soybean agriculture in the Midwest. These application methods, which include pesticide banding, fertilizer banding, and ridge tillage, could reduce polluted runoff. However, without any regulatory or economic incentives, these practices were not adopted by producers because they did not increase net returns (Iowa MSEA, 1995; Missouri MSEA, 1995).

Private research has been found to be responsive to regulations. Ollinger and Fernandez-Cornejo (1995) examined the effect of the Federal Insecticide, Fungicide, and Rodenticide Act on innovation in the agricultural chemical industry. They found the regulations resulted in the development of pesticides that were often less toxic and shorter lived than traditional pesticides (Ollinger and Fernandez-Cornejo, 1995).

## Summary

Research and development is an important part of a policy for reducing agricultural nonpoint-source pollution because it provides producers and society more efficient ways of meeting environmental goals. It may also, if directed toward monitoring technology, facilitate the eventual use of more efficient standards-based approaches to even nonpoint-source water quality improvement. Given the length of time it takes to develop and introduce new technology, R&D may require patience and a willingness to invest substantial private or public funds. However, since producers and private firms will necessarily underinvest in R&D for water quality improvements, the public sector will have to either carry out this research or provide producers and the private sector with incentives (through economic incentives or regulations) that result in efficient research investments. Price and regulatory signals that correctly reflect society's valuation of environmental problems can ensure that research is consistent with environmental goals.

[5] The research described above was not initiated specifically for the purpose of improving water quality.

Finally, it is important to recognize that while research is often viewed as one of the tools available for addressing water quality and other environmental problems (e.g., Clean Water Action Plan, USDA Water Quality Program), it cannot stand on its own as a tool to control water pollution. Instead, it is an extremely valuable component of other approaches that include performance or design incentives and standards. R&D cannot independently provide a solution to water quality problems because technology is only one component of water quality improvement. Even with the most efficient, environmentally friendly technology, producers have incentives to over- (under-) apply inputs that contribute to (inhibit) nonpoint-source pollution. Economically sound water quality policies will consider all aspects of the nonpoint problem to determine cost-effective solutions.

# Chapter 8

# Implications for Policy and Future Directions

*The previous chapters presented an extensive range of policy instruments that can be applied to agricultural sources of nonpoint-source pollution. Performance-based measures are generally infeasible at present because of the difficulty in observing nonpoint-source emissions and the information requirements placed on producers. The characteristics of nonpoint-source pollution (i.e., heterogeneous nature, variability, etc.) and the attractiveness of second-best policies (due to administrative costs, etc.) rule against a single policy tool. The most appropriate tool(s) for a particular problem is an empirical issue based on policy goals, local conditions, and costs of acquiring information. Research in areas such as offsite damages, implementation costs, and simulation models could enhance the performance of nonpoint pollution control policies.*

## Vehicle for Change

President Clinton's charge to chart a new course for nonpoint-source pollution policy recognizes that economic incentives, regulations (standards), education, and research all have a role to play in meeting clean water goals (EPA, USDA, 1998). To date, however, only some of these tools have actually been incorporated into State and Federal water quality programs. Programs designed to address agricultural nonpoint-source pollution have relied primarily on education, technical assistance, and short-term financial assistance. More recently, design standards have been incorporated into some State water quality programs.

This report has systematically presented an extensive range of policy instruments that can be applied to agricultural sources of nonpoint pollution. Unlike the existing economic literature on nonpoint policy tools in which a single study may consider only one or a limited set of nonpoint policy instruments, with varying assumptions across studies, this report has reviewed each policy tool using a unified framework. Consequently, a comparison of each instrument's strengths and weaknesses, with regard to economic efficiency and ease of administration, helps to identify which tools might best underpin a national agricultural water quality policy. In this chapter, we consider the full range of nonpoint instruments presented in this report and, taking the economic characteristics of each into account, we attempt to answer the following questions regarding implementation:

- Which instruments are most likely to achieve water quality goals at least cost, given the information that is likely to be available?
- Under what situations should each instrument be used?
- What information could a resource management agency obtain to improve the performance of the tools?

None of these questions implies that a single instrument or combination of instruments is best. Instead, the most appropriate instrument(s) is best determined case by case due to the heterogeneous nature of nonpoint pollution. At present, a comprehensive empirical assessment of different policy options does not exist. However, the limited economic literature providing empirical comparisons of some instruments is addressed in this chapter. Before assessing policy tools, however, we first review why policies for cost-effective nonpoint-source pollution control are so difficult to design.

## Complexities of Policy Design

Designing comprehensive policies for controlling nonpoint pollution consists of defining appropriate policy goals, choosing appropriate instruments, and setting these instruments at levels that will achieve the goals at least cost. Difficulties with each of these steps derive from the complex physical nature of nonpoint pollution.

Nonpoint emissions (runoff) cannot be measured at reasonable cost with current technologies because they are diffuse (i.e., they move off the fields in a great number of places) and are affected by random events such as weather, as is the process by which runoff is transported to a water body. This randomness narrows the way that policy goals with good economic properties are defined, and limits the types of policy tools that can be used to attain a cost-effective outcome. Finally, runoff depends on many site-specific factors. The more policies and goals are able to address these site-specific factors, the more efficient nonpoint policies will be.

## Assessment of Policy Goals

An economically efficient outcome is generally unattainable because policymakers seldom have information about economic damages. Instead, a cost-effective approach to nonpoint pollution control is typically preferred. A cost-effective outcome is an outcome in which policy goals are achieved at least cost. A variety of policy goals exist; however, the physical nature of nonpoint pollution limits the way in which the goals may be defined and also the economic properties of the goals. Apart from the economist's ideal outcome of economic efficiency, there are in general two types of policy goals: (1) physically based goals (water quality, runoff), and (2) input- and technology-based goals.

Physically based goals are limited in a number of ways. First, the random nature of the nonpoint process requires that these goals be set to attain a probability of occurrence of an outcome as opposed to a specific outcome (i.e., that a mean ambient pollution level be achieved, or that a particular ambient pollution level be achieved 95 percent of the time).

Second, the use of more stringent goals may not result in an expected reduction in damages. If not, then the adoption of more stringent goals (i.e., a 25-percent reduction in pollution levels as opposed to a 20-percent reduction) may actually make society worse off in its attempt to reduce pollution. Some techniques can be used to verify that physically based goals will reduce economic damages; however, the results may not always be conclusive.

Finally, the method of pollution control that achieves a physically based goal with greatest expected social net benefits (the sum of private pollution control costs plus the expected benefits of pollution reduction), known as the *economically preferred* method, will generally differ from the cost-effective method of achieving the same goal. The differences are due to risk effects that arise because the impact of each input on expected damages is not accounted for in the cost-effective outcome. For example, suppose the least-cost method of achieving a particular policy goal (method A) costs $50 and reduces expected damages by $100, for an expected net social gain of $50. Suppose method B also achieves the same goal, but at a cost of $60 and a reduction in expected damages by $120, for an expected net social gain of $60. In this case, method B is socially preferred to method A, even though method A achieves the goal at least cost. However, since damages often remain unknown and the economically preferred and cost-effective methods do not generally coincide, it will not be possible to identify the economically preferred method beforehand. Thus, the notion of cost-effectiveness is limited when policy goals are defined in terms of physical measures.

Input- and technology-based goals offer a practical alternative to physically based goals. For example, instead of designing policies to reduce mean nitrogen loadings, the goal may be a specified reduction in nitrogen fertilizer application rates. Such goals give policymakers more direct control over the factors that determine the distribution of outcomes, and can be chosen to ensure both a reduction in expected damages and an expected improvement in water quality. In addition, these goals can be set such that the cost-effective outcome is preferred to outcomes that achieve the goals at higher cost (i.e., the cost-effective and economically preferred outcomes may coincide). Finally, these goals can be set deterministically, making it easier to verify whether or not the goals are met. In contrast, it may take years to obtain a large enough sample to determine if probabilistic ambient water quality goals are achieved.

# Comprehensive Assessment of Policy Tools

## Performance-Based Instruments Face Insurmountable Problems

Performance-based instruments include those instruments based on the environmental outcomes of producer actions, such as runoff and ambient pollution levels. However, runoff-based instruments are not feasible since runoff cannot be accurately monitored with current technology.

Ambient-based instruments are (seemingly) advantageous because ambient pollution can be monitored (although at potentially high costs) without the resource management agency having to observe the actions of each producer. However, there are several difficulties associated with using ambient-based instruments. For example, ambient-based instruments can be designed to achieve an efficient or cost-effective outcome only under highly restrictive conditions, such as when producers are risk-neutral and producers and the resource management agency share the same expectations about the nonpoint process. This limitation is due to the complex, random nature of the nonpoint pollution process. Other limitations arise because ambient-based instruments depend on group performance. For these instruments to be effective, producers must be able to evaluate how their actions and the actions of others affect ambient pollution levels. Given the large numbers of nonpoint polluters that may exist within a region, and without concerted public sector R&D to resolve monitoring and forecasting technical problems, such instruments are likely to be too complex and information-intensive for producers to obtain all the required information and make accurate evaluations. In that case, producers will receive incorrect incentives from ambient-based instruments.

The resource management agency also has significant informational requirements in setting ambient-based instruments at appropriate levels because to do so requires that the agency understand how producers evaluate the impacts of their decisions on water quality. In other words, the agency must understand each producer's belief structure about the nonpoint process. This information is either not likely to be available, or is likely to be difficult and expensive to obtain.

## Feasible Policies Are Based on Observable Components of Production

If performance-based instruments are not viable instruments for controlling nonpoint-source pollution, design-based instruments are the only potential recourse. Design-based instruments are based on observable aspects of production such as input use or technology choice. In addition, ex ante performance measures such as expected runoff (defined as the level of runoff expected to result from specific production choices and calculated with the use of a runoff model) are included in the set of design bases.

### *Choice of base*

As pointed out in chapters 3 and 4, efficiency requires that design-based instruments be site-specific and applied to each variable input and technology choice. However, efficiency is not likely to be attainable, nor may it be desirable with high administrative (i.e., monitoring and enforcement) costs. Instead, second-best policies, based on a limited set of inputs or on expected runoff and applied uniformly across producers operating in a particular region, may be preferred.

First, consider expected runoff as an instrument base. This base is closer to the externality (pollution) than individual production decisions, allowing producers to remain somewhat flexible in how they control runoff. Producers are able to benefit from their specialized knowledge, to the extent that this knowledge can be captured by a model used to calculate expected runoff levels. In addition, expected runoff-based instruments have an "incentive effect," inducing producers to seek or to demand better technologies.

Expected runoff-based instruments also have a number of important drawbacks. First, the random nature of the nonpoint pollution process limits the types of outcomes that can be attained using expected runoff-based instruments. For example, the only cost-effective outcome that can be achieved with such an instrument is one designed to achieve a mean runoff goal. In addition, the use of each input and each technology choice must be monitored in order to apply the model to determine expected runoff levels. The administration costs are therefore not likely to be significantly reduced relative to other second-best instruments. Finally, producers are forced to use the resource management agency's expectations, as defined by the model, even though their own expectations may actually differ. The Universal Soil Loss Equation is the only model that might currently be accepted as a tool for predicting the runoff of a pollutant (Wischmeier and Smith, 1978). It has been used to assess eligibility for USDA programs, and for enforcing conservation compliance.

Alternatively, second-best, design-based instruments could be applied to a limited (truncated) set of inputs and/or technologies, and the instruments could be applied uniformly within a region. Second-best, design-based instruments could also be designed with limited information on the part of the resource management agency to help control administration costs. Such instruments may be effective in controlling nonpoint pollution if the inputs/technologies chosen as bases are highly correlated with water quality.

### *Design-based incentives vs. standards*

For a given instrument base, economic incentives or standards can be used to achieve identical policy goals. However, use of each instrument type will likely have different consequences for farm profitability by location. Distributional disparities will be greater the greater the heterogeneity of land, the more uniformly instruments are applied across a region, and the more uncertainty the resource management agency has about site-specific information when designing policies. In general, incentives provide more flexibility than standards because producers are free to adjust their production practices to take advantage of personal knowledge and to react to changing market conditions.

Incentives and standards will also have different administrative aspects. The information required by the resource management agency in setting design-based standards and incentives is very similar. However, monitoring may be easier for incentives that can be applied through existing markets. For example, a uniform fertilizer tax can be implemented as a sales tax whereas a fertilizer standard requires that each production site be monitored for fertilizer use. Design taxes have the additional advantage of generating revenue. This revenue could be used to support the administration of the water quality policy, to fund supporting programs such as education and research, or to retire marginal land. For example, a sales tax on fertilizer in Iowa was used to support nutrient management programs in that State (Mosher, 1987). While the tax rate is currently too low to affect behavior, research and education efforts may be increasing the efficiency of fertilizer use. That the nitrogen fertilizer application rate on corn is much lower in Iowa than for the other Corn Belt States is circumstantial evidence that research and education are having an effect (USDA, NASS-ERS, 1996).

Shortle and Dunn (1986) compared input standards, input incentives, expected runoff standards, and expected runoff incentives designed to achieve an efficient solution when asymmetric information exists. Ignoring transaction costs, they found that appropriately specified input incentives generally outperform input standards and expected runoff incentives and standards, given the characteristics of nonpoint source pollution and the information typically available to a resource management agency. These results, however, do not necessarily carry over to the case of multiple farms and/or second-best policies, where administrative costs are considered.

It is not possible to make a general statement about the relative performance of incentives and standards in a world with asymmetric information and second-best policies. Instead, there are situations in which each is preferred.[1] Similar conclusions can be made about the application of uniform policies across heterogeneous land. In general, each situation must be assessed individually.

Miltz, Braden, and Johnson (1988) compared uniform expected runoff standards and uniform expected runoff taxes. These instruments were compared in the context of soil erosion, where the Universal Soil Loss Equation and sediment delivery coefficients were used to estimate sediment discharge to waterways. They found that uniform discharge standards were superior to the uniform tax in achieving least-cost control if there were a strong correlation between the delivery coefficient and abatement cost. Otherwise, the tax is superior. For example, fields along a river on flat land would have higher delivery coefficients and lower ero-

[1] Weitzman (1974) examined price and quantity policies under asymmetric information and showed that, in cases where the marginal cost curve is nearly flat, an error in setting a tax could result in large deviations from the desired result, making standards the preferred instrument. Alternatively, when the benefit function is closer to being linear, price-based policies are superior. However, Malcomson (1978) showed that reliance on such simplistic criteria might result in the choice of incorrect policy tools. Similarly, Stavins (1996) showed the choice to be more complex when the uncertainty associated with the benefits and costs of pollution control are correlated.

sion rates than fields on hilly, upland areas away from the river. The marginal costs of reducing erosion are lower on the upland fields. A uniform tax would provide a greater incentive to reduce erosion on upland fields that may be contributing little to sediment in the river. A uniform standard would provide greater erosion control at least cost. Russell (1982) came to a similar conclusion when comparing similar instruments. Which tool is superior depends on the characteristics of the region, the size of the pollutant source, and the marginal cost of abatement.

Helfand and House (1995) found uniform input taxes to result in lower welfare costs than input standards to meet a desired water quality goal. These results held for taxes and standards applied to all inputs contributing to pollution, and also for the case in which only a single input was targeted.

Lichtenberg (1992) found that standards may be preferable to incentives when a specific input reduction goal is desired. For example, a standard would be preferred in a situation where a particular chemical is clearly detrimental to water quality and application rates need to be limited or the chemical banned from use. Setting a tax to optimally meet an input reduction goal requires knowledge of the farm-specific demand for that input. Such information is not likely to be available to a resource management agency. Design standards, or limits on input use, would be much easier to implement in this case, even though the distributive properties might be poor. Other examples where design standards might be preferred include chemigation (using irrigation equipment to apply chemicals along with water), chemical use on sandy soils, the use of vegetative buffers, and animal waste storage and use.

## Other Instruments Provide A Supportive Role

### *Education*

As shown in chapter 6, education by itself cannot achieve cost-effective water quality control, although it has proven valuable in support of other approaches. For example, Bosch, Cook, and Fuglie (1995) found that education enhanced the performance of a regulation requiring nitrogen testing in Nebraska. The regulation was more effective than education and cost-sharing in promoting adoption. However, producers did not use the information provided by the testing properly unless they received some educational assistance. Education and short-term cost-sharing accelerate the adoption process by providing producers with the means to acquire management skills and overcome short-term risks of new practices. Standards and economic incentives set the stage for producers to change management practices, but adoption and continued use is a multi-stage process that can fail at any of a number of points. Education can help overcome many of these constraints.

Education can also be an inexpensive way of improving the efficiency of input use under current technologies. To the extent that inefficient use of inputs is a source of water quality degradation, improving the management skills of producers enhances both net returns and environmental quality.

### *Research and development*

As with education, research is best suited in a support role for all pollution control policies. Research can provide producers and society with more efficient ways of meeting environmental goals. New inputs and technologies can help producers respond to water quality policies at least cost, while better information, monitoring technology, and models can help resource management agencies design more efficient policies.

## Heterogeneous Nature of Nonpoint Pollution Suggests a Mixed Policy

The wide variety of water pollution problems from agriculture (nitrates in surface- and groundwater, soil erosion, pesticides in groundwater, animal waste) and differences in agriculture and hydrology across regions probably argure against the use of a single policy tool. Multiple instruments have a role when a single instrument is inefficient because of the characteristics of nonpoint source pollution (Braden and Segerson, 1993). In his study of price and quantity-based policies, Weitzman (1974) concluded that mixed policies may give the best results in some situations, depending on the characteristics of the polluters and receiving waters. In a review of pollution policy tools, Baumol and Oates conclude that "...effective policy requires a wide array of tools and a willingness to use each of them as it is required" (Baumol and Oates, 1979, pp. 230-231).

Abler and Shortle (1991) reviewed the merits of a variety of tools (including education, design standards, performance standards, input taxes, input subsidies, performance taxes, and research and development) for reducing agricultural nonpoint-source pollution. Using evaluation criteria based on both economic and administrative attributes, they could not identify a single dominant tool. Each had its strengths and weaknesses. Which tools are actually preferred in a particular setting depends on the weights applied to the various attributes.

Shortle and Abler (1994) evaluated a mixed scheme consisting of marketable permits for polluting inputs combined with a tax on excess input use and a subsidy for returned permits. Such a scheme can be implemented without information on farm profits or offsite damage costs. This approach was generally shown to be superior to policies based solely on design incentives. Optimal implementation could still entail large administrative costs, but the mixed structure should offer opportunities for increased efficiency over input-based tax and license schemes that have been suggested as policies.

Conant, Duffy, and Holub (1993) studied how public policy can be fashioned to better address the perceived conflict between farm profitability and water quality practices. They examined how four different policies performed in achieving three different policy objectives in Iowa. The policies were (1) design standards for nutrient and pesticide management, (2) input taxes on nitrogen and pesticides, (3) technical assistance and cost-sharing for integrated crop management, and (4) research and education. The three policy objectives were to achieve maximum water quality improvement, to achieve greatest improvement in water quality consistent with maintaining farm profitability, and to achieve best overall improvement in both water quality and profitability. Each of the policies was examined, using models of representative farms in six Iowa counties, for impacts on farm profitability, nitrogen runoff, pesticide runoff, nitrate leaching, and pesticide leaching over a range of implementation levels. The major findings of the study, in terms of meeting policy objectives, are as follows:

- Taxation produced the greatest water quality benefit, but proved to be costly to producers.

- Water quality can be significantly improved without losses to farm profitability because of the profitability of alternative practices. Improvements to both can be achieved simultaneously, and in some cases, without high implementation and administrative costs.

- Changes in farming practices that might occur in response to a new policy are highly uncertain. However, this uncertainty affected the magnitude of the changes in water quality and farm profitability, not the direction.

- The impacts on water quality and profitability varied greatly across the State. This result implies that targeting different policies to different areas could improve efficiency.

## Institutional Issues

Coordination of water pollution control programs at the watershed level would promote economic efficiency. This suggests that policy tools should be tailored to the individual watershed wherever possible by State and local authorities. A watershed approach facilitates the identification of pollutants (and their source) that limit desired uses. Using best estimates of contributions from different sources, policy administrators can then select abatement goals and the instruments best suited to achieving those goals. Sunding (1996) shows how welfare losses from second-best policies can be reduced by making use of information that is relatively accessible at the regional level, including data on prices, crop yields, and production costs. These data can be used to tailor regulations on a regional basis to minimize losses in private welfare while achieving a particular environmental goal.

The Clean Water Act establishes a national goal for water quality, and this is reaffirmed by the recent Clean Water Action Plan (EPA, USDA, 1998). The Clean Water Action Plan also stresses the utility of a locally led watershed-based approach to water quality protection. A potential conflict arises because such an approach might not achieve national water quality goals, at least not efficiently. The Federal Government can mitigate this conflict by coordinating with the States to address transboundary problems. Without such intervention, the incentives for local jurisdictions to consider the transboundary implications of their own policies is lacking, and States may fail to meet their own water quality goals because of pollution from upstream activities. The Clean Water Act established a set of procedures for addressing such problems.

The Federal Government may be in a better position than local jurisdictions to support research on nonpoint-source pollution that has widespread benefits to States trying to implement their own programs. Examples would be the development of nonpoint-source pollution models and the development of new management practices. Similarly, by acting as a clearinghouse for scientific information on nonpoint pollution, the Federal Government can lower information costs to local jurisdictions.

In trying to balance Federal water quality goals with local ones, it might be necessary to establish minimum guidelines for water quality or for industry design standards. There might be some economies for a "leveling of the playing field" between jurisdictions. For instance, an industry may benefit if it does not have to meet 50 different sets of standards (Esty, 1996). Even though the guidelines may result in a higher cost to a local watershed for achieving its own water quality goals, the benefits to the economy as a whole outweigh these costs.

Minimum standards may also reduce the movement of more mobile industries to States with weaker environmental laws (Esty, 1996). There is some evidence that the location of large swine operations is at least partly due to differences in environmental regulations (Bacon, 1993; Hurt and Zering, 1993). The recently proposed Unified National Strategy for Animal Feeding Operations establishes national minimum standards for animal waste control for large animal feeding operations. One consequence is that States enacting more stringent laws to protect their water quality from animal waste are not hurt by the loss of business to States with less stringent laws. Minimum standards also protect States that might otherwise be slow to respond to environmental problems associated with polluting industries seeking a more favorable regulatory environment.

## References


Abler, D.G., and J.S. Shortle. 1991. "The Political Economy of Water Quality Protection from Agricultural Chemicals," *Northeastern Journal of Agricultural and Resource Economics* 21(1):53-60.

Abler, D.G., and J.S. Shortle. 1995. "Technology as an Agricultural Pollution Control Policy," *American Journal of Agricultural Economics* 77(1):20-32.

Abrahams, N., and J. Shortle. 1997. "The Value of Information and the Design of Environmental Policies for Agriculture," Selected paper presented at the American Agricultural Economics Association annual meetings, Toronto, Canada, July.

Adler, R.W. 1994. "Reauthorizing the Clean Water Act: Looking to Tangible Values," *Water Resources Bulletin* 30(5):799-807.

Anderson, J.L., F.G. Bergsrud, and T.M. Ahles. 1995. "Evaluation of the Farmstead Assessment System (FARM*A*SYST) in Minnesota," in *Clean Water-Clean Environment-21st Century, Vol. III: Practices, Systems, & Adoption*. Proceedings of the conference Clean Water-Clean Environment-21st Century, Working Group on Water Quality, USDA. March 5-8, Kansas City, MO, pp. 9-12.

Animal Confinement Policy National Task Force. 1998. *1998 National Survey of Animal Confinement Policies*. http://cherokee.agecon.clemson.edu/confine.htm.

Antle, J.M., and T. McGuckin. 1993. "Technological Innovation, Agricultural Productivity, and Environmental Quality," in Carlson, G.A., D. Zilberman, and J.A. Miranowski (eds.), *Agricultural and Environmental Resource Economics.* New York, NY: Oxford University Press.

Antweiler, R.C., D.A. Goolsby, and H.E. Taylor. 1995. "Nutrients in the Mississippi River," in Meade, R.H. (editor) *Contaminants in the Mississippi River.* USGS Circular 1133. U.S. Dept. Interior, U.S. Geological Survey.

Babcock, B.A., P.G. Lakshminarayan, J. Wu, and D. Zilberman. 1997. "Targeting Tools for the Purchase of Environmental Amenities," *Land Economics* 73(3):325-339.

Bacon, J.A. 1993. "OINK! Not North Carolina!" *Virginia Business* (June):55-56.

Barbash, J.E,. and E.A. Resek. 1995. *Pesticides in Ground Water: Current Understanding of Distribution and Major Influences*. U.S. Dept. Interior, U.S. Geological Survey.

Bartfeld, E. 1993. "Point-Nonpoint Source Trading: Looking Beyond Potential Cost Savings," *Environmental Law* 23(43):43-106.

Baumol, W.J., and W.E. Oates. 1979. *Economics, Environmental Policy, and the Quality of Life.* Englewood Cliffs, NJ: Prentice Hall.

Baumol, W.J., and W.E. Oates. 1988. *The Theory of Environmental Policy.* New York: Cambridge University Press.

Beach, E.D., and G.A. Carlson. 1993. "A Hedonic Analysis of Herbicides: Do User Safety and Water Quality Matter?" *American Journal of Agricultural Economics* 75(3):612-623.

Beavis, B., and I. Dobbs. 1987. "Firm Behaviour under Regulatory Control of Stochastic Environmental Wastes by Probabilistic Constraints," *Journal of Environmental Economics and Management* 14:112-127.

Beavis, B., and M. Walker. 1983. "Achieving Environmental Standards with Stochastic Discharges," *Journal of Environmental Economics and Management* 10:103-111.

Bishop, R. 1994. "A local agency's approach to solving the difficult problem of nitrate in the groundwater," *Journal of Soil and Water Conservation* 49(2 ss):82-84.

Boesch, D.F., D.M. Anderson, R.A. Horner, S.E. Shumway, P.A. Tester, and T.E. Whitledge. 1997. *Harmful Algal Blooms in Coastal Waters: Options for Prevention, Control and Mitigation.* Decision Analysis Series No. 10. U.S. Dept. of Commerce, NOAA Coastal Ocean Program.

Bohm, P., And C. Russell. 1985. "Comparative Analysis of Alternative Policy Instruments," in Kneese, A.Y., and J.L. Sweeny (eds.) *Handbook of Natural Resource Economics, Vol. 1*. New York, NY: Elsevier Science Publishing Co.

Bosch, D.J., N.L. Powell, and F.S. Wright. 1992. "An Economic Comparison of Subsurface Microirrigation with Center Pivot Sprinkler Irrigation for Row Crop

Production," *Journal of Production Agriculture* 5(4):431-437.

Bosch, D., Z. Cook, and K. Fuglie. 1995. "Voluntary Versus Mandatory Agricultural Policies to Protect Water Quality: Adoption of Nitrogen Testing in Nebraska," *Review of Agricultural Economics* 17(2):13-24.

Braden, J.B., and N. Matsueda. 1997. "Environment, Federalism, and Economics: How do Three Become One?" Paper presented at Resource Policy Consortium, Washington, DC, June 5.

Braden, J.B., and K. Segerson. 1993. "Information Problems in the Design of Nonpoint-Source Pollution Policy," in Russell, C.S., and J.F. Shogren (eds.) *Theory, Modeling, and Experience in the Management of Nonpoint-Source Pollution*. Boston, MA: Kluwer Academic Publishers, pp. 1-36.

Bull, L., And C. Sandretto. 1995. "The economics of agricultural tillage systems," *Farming for a Better Environment*. Ankeny, IA: Soil and Water Conservation Society of America, pp. 35- 40.

Cabe, R., and J. A. Herriges. 1992. "The Regulation of Non-Point-Source Pollution Under Imperfect and Asymmetric Information," *Journal of Environmental Economics and Management* 22:134-146.

Camboni, S.M., and T.L. Napier. 1994. "Socioeconomic and Farm Structure Factors Affecting Frequency of Use of Tillage Systems." Invited paper presented at the Agrarian Prospects III symposium, Prague, Czech Republic, September.

Carson, R.T., and R.C. Mitchell. 1993. "The Value of Clean Water: The Public's Willingness to Pay for Boatable, Fishable, and Swimmable Water Quality," *Water Resources Research* 29(7):245-254.

Centers for Disease Control and Prevention. 1995. *Morbidity and Mortality Weekly Report.* June 16.

Centers for Disease Control and Prevention. 1996. *Surveillance for Waterborne-Disease Outbreaks—United States, 1993-1994*. 45(SS-1).

Centner, T.J. 1990. "Blameless Contamination: New state legislation regulating liability for agricultural chemicals in groundwater," *Journal of Soil and Water Conservation* 44(2):216-220.

Clark, E.H., II, J.A. Haverkamp, and W. Chapman. 1985. *Eroding Soils: The Off-Farm Impacts.* Washington, DC: The Conservation Foundation.

Coehlo, M.B., and E.T. Kornegay (editors). 1996. *Phytase in Animal Nutrition and Waste Management: A BASF Reference Manual 1996.* Mt. Olive, NJ: BASF Corp.

Conant, C.K., M.D. Duffy, and M.A. Holub. 1993. *Tradeoffs between Water Quality and Profitability in Iowa Agriculture*. University of Iowa, Iowa City, IA,

Cooper, J.C., and R.W. Keim. 1996. "Incentive Payments to Encourage Farmer Adoption of Water Quality Protection Practices," *American Journal of Agricultural Economics* 78(1):54-64.

Council for Agricultural Science and Technology (CAST). 1996. *Integrated Animal Waste Management*. Task Force Report No. 128, November.

Crutchfield, S.R., P.M. Feather, and D.R. Hellerstein. 1995. *The Benefits of Protecting Rural Water Quality: An Empirical Analysis.* AER-701, U.S. Dept. Agr., Econ. Res. Serv., January.

Denbaly, M., and H. Vroomen. 1991. *Elasticities of Fertilizer Demands for Corn in the Short and the Long Run.* Staff Report AGES 9137, U.S. Dept. Agr., Econ. Res. Serv., Aug.

Dicks, M.R. 1986. "What Will It Cost Farmers to Comply with Conservation Compliance?" *Agricultural Outlook*, No. 561.

Easter, K.W. 1993. "Differences in the Transaction Costs of Strategies to Control Agricultural Offsite and Undersite Damages," in Russell, C.S., and J.F. Shogren (eds.), *Theory, Modeling, and Experience in the Management of Nonpoint-Source Pollution.* Boston, MA: Kluwer Academic Publishers. pp. 37-68.

Eiswerth, M.E. 1993. "Regulatory/Economic Instruments for Agricultural Pollution: Accounting for Input Substitution," in Russell, C.S., and J.F. Shogren (eds), *Theory, Modeling, and Experience in the Management of Nonpoint-Source Pollution.* Boston, MA: Kluwer Academic Publishers. pp. 69-90.

Engler, Reto. 1993. "Lists of Chemicals Evaluated for Carcinogenic Potential." Memorandum, Office of

Prevention, Pesticides, and Toxic Substances, Environmental Protection Agency, Aug. 31.

Environmental Law Institute. 1997. *Enforceable State Mechanisms for the Control of Nonpoint Source Water Pollution*. Washington, DC.

Environmental Law Institute. 1998. *An Almanac of Enforceable State Laws to Control Nonpoint Source Water Pollution*. Washington, DC.

Ervin, D.E. 1995. "A New Era of Water Quality Management in Agriculture: From Best Management Practices to Watershed-Based Whole Farm Approaches?" *Water Resources Update* 101:18-28.

Ervin, C.A., and D.E. Ervin. 1982. "Factors Affecting the Use of Soil Conservation Practices: Hypotheses, Evidence, and Policy Implications," *Land Economics* 58(3):277-292.

Esseks, J.D., and S.E. Kraft. 1993. "Opinions of Conservation Compliance Held by Producers Subject to It." Report for the American Farmland Trust. February.

Esty, D.C. 1996. " Revitalizing Environmental Federalism," *Michigan Law Review* 95:570-653.

Feather, P., and D. Hellerstein. 1997. "Calibrating Benefit Function Transfer to Assess the Conservation Reserve Program," *American Journal of Agricultural Economics* 79(1):151-162.

*Federal Register.* 1998. "Unified National Strategy for Animal Feeding Operations," 63(182):50192-50209.

Fernandez-Cornejo, J., S. Jans, and M. Smith. 1998. "Issues in the Economics of Pesticide Use in Agriculture: A Review of the Empirical Evidence." *Review of Agricultural Economics*. 20(Fall/Winter): 462-488.

Fort, D.D. 1991. "Federalism and the Prevention of Groundwater Contamination," *Water Resources Research* 27(11):2811-2817.

Fox, G., A. Weersink, G. Sarwar, S. Duff, and B. Deen. 1991. "Comparative Economics of Alternative Agricultural Production Systems: A Review," *Northeastern Journal of Agricultural and Resource Economics* 20(1):124-142.

Franco, J., S. Schad, and C.W. Cady. 1994. "California's experience with a voluntary approach to reducingnitrate contamination of groundwater: The Fertilizer Research and Education Program (FREP)," *Journal of Soil and Water Conservation* 49(2 ss):76-81.

Freeman, A.M., III. 1982. *Air and Water Pollution Control: A Benefit-Cost Assessment.* New York: John Wiley and Sons.

Freeman, A.M., III. 1993. *The Measurement of Environmental and Resource Values: Theory and Method.* Washington, DC: Resources for the Future.

Fuglie, K.O., and D.J. Bosch. 1995. "Economic and Environmental Implications of Soil Nitrogen Testing: A Switching-Regression Analysis," *American Journal of Agricultural Economics* 77(4):891-900.

Fuglie, K., N. Ballenger, K. Day, C. Klotz, M. Ollinger, J. Reilly, U. Vasavada, and J. Yee. 1996. *Agricultural Research and Development: Public and Private Investments Under Alternative Markets and Institutions*. AER-735. U.S. Dept. Agr., Econ. Res. Serv., May.

Goolsby, D.A., and W.A. Battaglin. 1993. "Occurrence, Distribution, and Transport of Agricultural Chemicals in Surface Waters of the Midwestern United States," in Goolsby, D.A., L.L. Boyer, and G.E. Mallard (eds.), *Selected Papers on Agricultural Chemicals in Water Resources of the Midcontinental United States*. Open-File Report 93-418. U.S. Dept. Interior, U.S. Geological Survey, pp. 1-25.

Goolsby, D.A., and W.A. Battaglin. 1997. "Effects of Episodic Events on the Transport of Nutrients to the Gulf of Mexico," in U.S. EPA, *Proceedings of the First Gulf of Mexico Hypoxia Management Conference*, December 5-6, 1995, Kenner, LA. EPA-55-R-97-001. Gulf of Mexico Program Office, pp. 144-145.

Goolsby, D.A., E.M. Thurman, M.L. Pomes, M. Meyer, and W.A. Battaglin. 1993. "Occurrence, Deposition, and Long Range Transport of Herbicides in Precipitation in the Midwestern and Northeastern United States," in Goolsby, D.A., L.L. Boyer, and G.E. Mallard (eds.), *Selected Papers on Agricultural Chemicals in Water Resources of the Midcontinental United States*. Open-File Report 93-418. U.S. Dept. Interior, U.S. Geological Survey, pp. 75-88.

Gould, B.W., W.E. Saupe, and R.M. Klemme. 1989. "Conservation Tillage: The Role of Farm and Operator

Characteristics and the Perception of Soil Erosion," *Land Economics* 85(2):167- 182.

Graham, E. 1997. "TMDL's and Effluent Trading: The Key to Healthy Watersheds?" Remarks before the conference "TMDL and Effluent Trading: The Key to Healthy Watersheds?" Chesapeake Water Environment Association and the American Water Resources Association, College Park, MD, Nov.

Hansen, L.G. 1998. "A Damage Based Tax Mechanism for Regulation of Non-Point Emissions," *Environmental and Resource Economics* 12(1):99-112.

Hayami, Y., and V. Ruttan. 1985. *Agricultural Development: An International Perspective.* Baltimore, MD: Johns Hopkins University Press.

*Health and Environment Digest.* 1994. "*Cryptosporidium* and Public Health," 8(8):61-63.

Helfand, G.E., and B.W. House. 1995. "Regulating Nonpoint Source Pollution Under Heterogeneous Conditions," *American Journal of Agricultural Economics* 77(4):1024-1032.

Herriges, J.R., R. Govindasamy, and J. Shogren. 1994. "Budget-Balancing Incentive Mechanisms," *Journal of Environmental Economics and Management* 27:275-285.

Hickman, J.S., C.P. Rowell, and J.R. Williams. 1989. "Net returns from conservation compliance for a producer and landlord in northeastern Kansas," *Journal of Soil and Water Conservation* 44(5):532-534.

Higgins, E.M. 1995. *The Water Quality Incentive Program: The Unfulfilled Promise*. Walthill, NE: Center for Rural Affairs.

Hoag, D.L., and H.A. Holloway. 1991. "Farm Production Decisions Under Cross and Conservation Compliance," *American Journal of Agricultural Economics* 73(1):184-193.

Hoag, D.L., and J.S. Hughes-Popp. 1997. "Theory and Practice of Pollution Credit Trading in Water Quality Management," *Review of Agricultural Economics* 19(2):252-262.

Hoban, T.J., and R.C. Wimberley. 1992. "Farm Operators' Attitudes About Water Quality and the RCWP." In U.S. EPA, *Seminar Publication: The National Rural Clean Water Program Symposium.* Proceedings of the conference, *10 Years of Controlling Agricultural Nonpoint Source Pollution: The RCWP Experience*, September 13-17, Orlando, FL. EPA/625/R-92/006. Office of Water, pp. 247-254.

Hopkins, J., G. Schnitkey, and L. Tweeten. 1996. "Impacts of Nitrogen Control Policies on Crop and Livestock Farms at Two Ohio Farm Sites," *Review of Agricultural Economics* 18:311- 324.

Horan, R.D. 1999. "The 'Satisficing' Approach to Nonpoint Pollution Control: An Economic Assessment of Alternative Policy Goals," ERS Working Paper, USDA.

Horan, R.D., J.S. Shortle, and D.G. Abler. 1998a. "Ambient Taxes under Heterogeneous Expectations and Risk Aversion." Selected paper presented at the American Agricultural Economic Association annual meeting, Salt Lake City, Utah, August 2-5.

Horan, R.D., J.S. Shortle, and D.G. Abler. 1998b. "Ambient Taxes when Polluters Have Multiple Decisions," *Journal of Environmental Economics and Management* 36(2):186-199.

Hrubovcak, J., M. LeBlanc, and B.K. Eakin. 1995. *Accounting for the Environment in Agriculture.* TB-1847. U.S. Dept. Agr., Econ. Res. Serv., Oct.

Hurt, C., and K. Zering. 1993. "Hog Production Booms in North Carolina: Why There? Why Now?" *Purdue Agricultural Economics Report* (August):11-14.

Iowa Dept. of Natural Resources. 1990. *Livestock Waste Control Programs of Ten Midwest and Western States*. Des Moines, IA, Nov.

Iowa Management Systems Evaluation Area Project. 1995. *Impact of Farming Practices on Water Quality in Iowa.* March.

Juranek, D. 1995. "*Cryptosporidiosis*: Source of Infection and Guidelines for Prevention," *Clinical Infectious Diseases* 21(Suppl 1):57-61.

Just, R.E., D.L. Hueth, and A. Schmitz. 1982. *Applied Welfare Economics and Public Policy.* Englewood Cliffs, NJ: Prentice Hall.

Kellogg, R.L., M.S. Maizel, and D.W. Goss. 1992. *Agricultural Chemical Use and Ground Water Quality:*

*Where Are the Potential Problem Areas?* U.S. Dept. Agr., Soil Cons. Serv., Econ. Res. Serv., Coop. State. Res. Serv., and Nat. Cent. for Resource Innovations. Dec.

Knopman, D.S., and R.A. Smith. 1993. "20 Years of the Clean Water Act," *Environment* 35(1):17-20, 35-51.

Knox, D., G. Jackson, and E. Nevers. 1995. *Farm A Syst Progress Report 1991-1994.* University of Wisconsin Cooperative Extenstion.

Kolpin, D.W., M.R. Burkart, and E.M. Thurman. 1994. *Herbicides and Nitrate in Near-Surface Aquifers in the Midcontinental United States, 1991.* Water Supply Paper 2413. U.S. Geological Survey.

Kraft, S.E., C. Lant, and K. Gillman. 1996. "WQIP: An assessment of its chances for acceptance by farmers," *Journal of Soil and Water Conservation* 51(6): 494-498.

Larson, D., G. Helfand, and B. House. 1996. "Second-Best Tax Policies to Reduce Nonpoint Source Pollution," *American Journal of Agricultural Economics* 78(4):1108-1117.

Lee, J.G., R.D. Lacewell, and J.W. Richardson. 1991. "Soil Conservation or Commodity Programs: Tradeoffs During the Transition to Dryland Crop Production," *Southern Journal of Agricultural Economics* 23(1): July.

Lee, L.K., and R.L. Leonard. 1990. "Farmer liability for pesticide contamination of groundwater in Connecticut," *Journal of Soil and Water Conservation* 44(2):221-222.

Lester. J.P. 1994. "A New Federalism? Environmental Policy in the States," in Vig, N.J., and M.E. Kraft (eds.), *Environmental Policy in the 1990's: Toward a New Agenda.* Washington, DC: CQ Press, pp. 51-67.

Letson, D., and N. Gollehon. 1996. "Confined Animal Production and the Manure Problem," *Choices* Third Quarter:18-24.

Letson, D., S. Crutchfield, and A. Malik. 1993. *Point-Nonpoint Source Trading for Managing Agricultural Pollutant Loadings.* AER-674. U.S. Dept. Agr., Econ. Res. Serv. Sept.

Lettenmaier, D.P., E.R. Hooper, C. Wagoner, and K.B. Faris. 1991. "Trends in Stream Quality in the Continental United States, 1978-1987," *Water Resources Research* 27(3):327-339.

Lichtenberg, E. 1992. "Alternative Approaches to Pesticide Regulations," *Northeastern Journal of Agricultural and Resource Economics* 21(2):83-92.

Lichtenberg, E., and B.V. Lessley. 1992. "Water quality, cost-sharing, and technical assistance: Perceptions of Maryland farmers," *Journal of Soil and Water Conservation* 47(3):260-263.

Lichtenberg, E., I. E. Strand, and B.V. Lessley. 1993. "Subsidizing Agricultural Nonpoint-Source Pollution Control: Targeting Cost Sharing & Technical Assistance," in Russell, C.S., and J.F. Shogren (eds.), *Theory, Modeling, and Experience in the Management of Nonpoint-Source Pollution.* Boston, MA: Kluwer Academic Publishers. pp. 305-328.

Lohman, L.C., J.G. Milliken, and W.S. Dorn. 1988. *Estimating Economic Impacts of Salinity of the Colorado River.* U.S. Dept. Interior, Bureau of Reclamation. Feb.

Lynne, G.D., J.S. Shonkwiler, and L.R. Rola. 1988. "Attitudes and Farmer Conservation Behavior," *American Journal of Agricultural Economics* 70(1):12-19.

MacKenzie, W.R., N.J. Hoxie, M.E. Proctor, M.S. Gradus, K.A. Blair, D.E. Peterson, J.J. Kazmeirczak, D.G. Addiss, K.R. Fox, J.B. Rose, and J.P. Davis. 1994. "A Massive Outbreak in Milwaukee of *Cryptosporidium* Infection Transmitted through the Public Water Supply," *The New England Journal of Medicine* 331(3):161-167.

Malcomson, J.M. 1978. "Prices vs. Quantities: A Critical Note on the Use of Approximations," *Review of Economic Studies* 68:203-207.

Malik, A.S., B.A. Larson, and M. Ribaudo. 1992. *Agricultural Nonpoint Source Pollution and Economic Incentive Policies: Issues in the Reauthorization of the Clean Water Act.* Staff Report AGES 9229. U.S. Dept. Agr., Econ. Res. Serv. Nov.

Malik, A.S., B.A. Larson, and M. Ribaudo. 1994. "Economic Incentives for Agricultural Nonpoint Source Pollution Control," *Water Resources Bulletin* 30(3):471-480.

Meisinger, J.J. 1984. "Evaluating Plant-Available Nitrogen in Soil-Crop System," *Nitrogen in Crop Production*. Madison, WI: Soil Science Society of America.

Meisinger, J.J., and G.W. Randall. 1991. "Estimating Nitrogen Budgets for Soil-Crop Systems," *Managing Nitrogen for Groundwater Quality and Farm Profitability*. Madison, WI: Soil Science Society of America, Madison, WI.

Miceli, T.J., and K. Segerson. 1991. "Joint Liability in Torts: Marginal and Infra-Marginal Efficiency," *International Review of Law and Economics* 11:235-249.

Miltner, R.J., D.B. Baker, T.F. Speth, and C.A. Fronk. 1989. "Treatment of seasonal pesticides in surface waters," *American Water Works Journal* 81(1):43-52.

Miltz, D., J.B. Braden, and G.V. Johnson. 1988. "Standards Versus Prices Revisited: The Case of Agricultural Non-Point Source Pollution," *Journal of Agricultural Economics* 39:360-368.

Miranowski, J. 1978. "Economic Implications of 208 Planning on the Agricultural Economy," in Baker, M. (ed.), *The Economic Impact of Section 208 Planning on Agriculture.* Publication No. 86, Great Plains Agricultural Council. Lincoln, NE: University of Nebraska.

Missouri Management Systems Evaluation Area Project. 1995. *A Farming Systems Water Quality Report*. Research and Education Report.

Mlot, C. 1997. "The Rise in Toxic Tides," *Science News.* September 27:202-204.

Morandi, L. 1989. *State Groundwater Protection Policies: A Legislator's Guide.* Washington, DC: National Conference of State Legislatures.

Moreau, D.H. 1994. "Water Pollution Control in the United States: Policies, Planning and Criteria," *Water Resources Update* 94:2-23.

Moreau, R., and J. Strasma. 1995. *Measuring the Benefits and Costs of Voluntary Pollution Prevention Programs in the Agricultural Sector Under Three Alternative Concepts—Averting Expenditures, Willingness-to-Pay, and Avoidance Costs: A Benefit-Cost Analysis of the Farm Assessment System (Farm*A*Syst) as Implemented in Nine Parishes in Louisiana.* Unpublished paper, University of Wisconsin.

Moschini, G., and H. Lapan. 1997. "Intellectual Property Rights and the Welfare Effects of Agricultural R&D," *American Journal of Agricultural Economics* 79(1):1229-1242.

Mosher, L. 1987. "How to deal with groundwater contamination?," *Journal of Soil and Water Conservation* 42(5):333-335.

Mueller, D.K., and D.R. Helsel. 1996. *Nutrients in the Nation's Water - Too Much of a Good Thing?* U.S. Geological Survey Circular 1136. U.S. Dept. Interior, U.S. Geological Survey.

Mueller, D.K., P.A. Hamilton, D.R. Helsel, K.J. Hitt, and B.C. Ruddy. 1995. *Nutrients in Ground Water and Surface Water of the United States - An Analysis of Data Through 1992.* Water-Resources Investigations Report 95-4031. U.S. Dept. Interior, U.S. Geological Survey.

Musser, W.C., J.S. Shortle, K. Kreahling, B. Roach, W.C. Huang, D. Beegle, and R.H. Fox. 1995. "An Economic Analysis of the Pre-Sidedress Soil Nitrogen Test for Pennsylvania Corn Production," *Review of Agricultural Economics* 17:25-35.

Napier, T.L., and D.E. Brown. 1993. "Factors affecting attitudes toward groundwater pollution among Ohio farmers," *Journal of Soil and Water Conservation* 48(5):432-438.

Napier, T.L., and S.M. Camboni. 1993. "Use of conventional and conservation practices among farmers in the Scioto River Basin of Ohio," *Journal of Soil and Water Conservation* 48(3):231- 237.

Natural Resource Defense Council. 1998. *America's Animal Factories: How States Fail to Prevent Pollution from Livestock Waste.* http://www.nrdc.org/nrdc/nrdcpro/factor/.

Nelson, M.C., and W.D. Seitz. 1979. "An Economic Analysis of Soil Erosion Control in a Watershed Representing Corn Belt Conditions," *North Central Journal of Agricultural Economics* 1(2):July.

Nielsen, E.G., J.A. Miranowski, and M.J. Morehart. 1989. *Investments in Soil Conservation and Land Improvements.* AER-601. U.S. Dept. Agr., Econ. Res. Serv. Jan.

Norris, P.E., and S. S. Batie. 1987. "Virginia Farmers' Soil Conservation Decisions: An Application of Tobit Analysis," *Southern Journal of Agricultural Economics* 19:79-90.

Northeast Regional Agricultural Engineering Service. 1996. *Animal Agriculture and the Environment: Nutrients, Pathogens, and Community Relations.* Proceedings from the Animal Agriculture and the Environment North American Conference. Ithaca, NY: NRAES Cooperative Extension. Dec.

Nowak, P.J. 1987. "The Adoption of Agricultural Conservation Technologies: Economic and Diffusion Explanations," *Rural Sociology* 52(2): 208-220.

Nowak, Peter J. 1991. "Why Farmers Adopt Production Technology," in *Crop Residue Management for Conservation*, proceedings of a national conference, Soil and Water Conservation Society.

Nowak, P.J., G. O'Keefe, C. Bennett, S. Anderson, and C. Trumbo. 1997. *Communication and Adoption Evaluation of USDA Water Quality Demonstration Projects*. Madison, WI: University of Wisconsin in cooperation with U.S. Department of Agriculture. Oct..

Oates, W.E., and R.M. Schwab. 1988. "Economic Competition Among Jurisdictions: Efficiency Enhancing or Distortion Inducing?" *Journal of Public Economics* 35:333-354.

Ollinger, M., and J. Fernandez-Cornejo. 1995. *Innovation, Regulation, and Market Structure in the U.S. Pesticide Industry.* AER-719. U.S. Dept. Agr., Econ. Res. Serv.

Ollinger, M. and J. Fernandez-Cornejo. 1998. "Innovation and Regulation in the Pesticide Industry," *Agricultural and Resource Economics Review* 27(1):15-27.

Olson, E. 1995. "You Are What You Drink . . ." Briefing paper, Natural Resources Defense Council, June.

Osborn, C.T., and P.P. Setia. 1988. "Estimating the Economic Impacts of Conservation Compliance: A Corn Belt Application." Selected paper presented at the American Agricultural Economics Association annual meeting, Knoxville, TN, Aug.

Padgitt, S. 1989. *Farmers' views on groundwater quality: Concerns, practices, and policy preferences.* Staff report. U.S. Office of Technology Assessment, Washington, DC, Feb.

Pait, A., A. DeSouza, and D. Farrow. 1992*.* *Agricultural Pesticide Use in Coastal Areas: A National Summary.* U.S. Dept. of Commerce, National Oceanic and Atmospheric Administration, Sept

Pease, J., and D. Bosch. 1994. "Relationships among farm operators' water quality opinions, fertilization practices, and cropland potential to pollute in two regions of Virginia," *Journal of Soil and Water Conservation* 49(5):477-483.

Peters, M., H. McDowell, and R. House. 1997. "Environmental and Economic Effects of Taxing Nitrogen Fertilizer." Selected paper presented at the annual meetings of the American Agricultural Economics Association, Toronto, Ontario, Canada, July.

Pimentel, D., D. Andow, R. Dyson-Hudson, D. Gallahan, S. Jacobson, M. Irish, S. Kroop, A. Moss, I. Schreiner, M. Shepard, T. Thompson, and B. Vinzant. 1991. "Environmental and Social Costs of Pesticides: A Preliminary Assessment," in Pimentel, D. (ed.), *Handbook of Pest Management in Agriculture Vol. I.* Boca Raton, FL: CRC Press, Inc. pp. 721-740.

Prato, T., and S. Wu. 1991. "Erosion, Sediment and Economic Effects of Conservation Compliance in an Agricultural Watershed," *Journal of Soil and Water Conservation* 46(3):211- 214.

Price, K.A. 1982. *Regional Conflict & National Policy*. Washington, DC: Resources for the Future.

Puckett, L.J. 1994. *Nonpoint and Point Sources of Nitrogen in Major Watersheds of the United States.* Water-Resources Investigations Report 94-4001. U.S. Dept. Interior, U.S. Geological Survey.

Rabalais, N.N., R.E. Turner, and W.J. Wiseman, Jr. 1997. "Hypoxia in the Northern Gulf of Mexico: Past, Present and Future," in EPA, *Proceedings of the First Gulf of Mexico Hypoxia Management Conference*, Dec. 5-6, 1995, Kenner, LA. EPA-55-R-97-001. Gulf of Mexico Program Office, pp. 25-36.

Reichelderfer, K. 1990. "Environmental Protection and Agricultural Support: Are Tradeoffs Necessary?" in K. Allen (ed.), *Agricultural Policies in a New Decade.* Washington, DC: Resources for the Future., pp. 201-230.

Revesz, R. 1992. "Rehabilitating Interstate Competition: Rethinking the "Race to the Bottom" Rationale for Federal Environmental Regulation," *New York University Law Review* 67:1210.

Ribaudo, M.O. 1986. *Reducing Soil Erosion: Offsite Benefits.* AER-561. U.S. Dept. Agr., Econ. Res. Serv., Sept.

Ribaudo, M.O. 1989. *Water Quality Benefits from the Conservation Reserve Program.* AER-606. U.S. Dept. Agr., Econ. Res. Serv., Feb.

Ribaudo, M.O., and A. Bouzaher. 1994. *Atrazine: Environmental Characteristics and Economics of Management.* AER-699. U.S. Dept. Agr., Econ. Res. Serv., Sept.

Ribaudo, M.O., and D. Hellerstein. 1992. *Estimating Water Quality Benefits: Theoretical and Methodological Issues.* TB-1808. U.S. Depart. Of Agr., Econ. Res. Serv., Sept.

Ribaudo, M.O., and R.A. Shoemaker. 1995. "The Effect of Feedgrain Program Participation on Chemical Use," *Agricultural and Resource Economic Review* 24(2):211-220.

Richardson, J.W., D.C. Gerloff, B.L. Harris, and L.L. Dollar. 1989. "Economic impacts of conservation compliance on a representative Dawson County, Texas farm," *Journal of Soil and Water Conservation* 44(5):527-531.

Russell, C.S. 1986. "A Note on the Efficiency Ranking of Two Second-Best Policy Instruments for Pollution Control," *Journal of Environmental Economics and Management* 13:13-17.

Russell, C.S., and W.J. Vaughan. 1982. "The National Recreational Fishing Benefits of Water Pollution Control," *Journal of Environmental Economics and Management* 9:328-354.

Ruttan, V.W., and Y. Hayami. 1989. "Induced Technical Change in Agriculture," in Capalbo, S.M., and J.M. Antle (eds.), *Agricultural Productivity: Measurement and Explanation.* Washington, DC: Resources for the Future.

Satchell, M. 1996. "Hog heaven–and hell," *U.S. News & World Report*, Jan. 22:55-59.

Segerson, K. 1988. "Uncertainty and Incentives for Nonpoint Pollution Control," *Journal of Environmental Economics and Management* 15:87-98.

Segerson, K. 1990. "Liability for Groundwater Contamination from Pesticides," *Journal of Environmental Economics and Management* 19:227-243.

Segerson, K. 1995. "Liability and Penalty Structures in Policy Design," in Bromley, D.W. (ed.), *The Handbook of Environmental Economics.* Cambridge, MA: Basil Blackwell, Ltd.

Shavell, S. 1987. "Liability Versus Other Approaches to the Control of Risk," in *Economic Analysis of Accident Law*, Cambridge, MA: Harvard University Press.

Shortle, J.S. 1984. "The Use of Estimated Pollution Flows in Agricultural Pollution Control Policy: Implications for Abatement and Policy Instruments," *Northeastern Journal of Agricultural and Resource Economics* 13:277-285.

__________. 1987. "Allocative Implications of Comparisons Between the Marginal Costs of Point and Nonpoint Source Pollution Abatement," *Northeastern Journal of Agricultural and Resource Economics* 16:17-23.

_________. 1990. "The Allocative Efficiency Implications of Water Pollution Abatement Cost Comparisons," *Water Resources Research* 26(5):793-797.

_________. 1995. "Environmental Federalism: The Case of U.S. Agriculture?" in Braden, J.R., H. Folmer, and T. Ulen, eds. *Environmental Policy with Economic and Political Integration: The European Union and the United States.* Cheltenham, UK: Edward Elgar.

Shortle, J.S., and D.G. Abler. 1994. "Incentives for Agricultural Nonpoint Pollution Control," in Graham-Tomasi, T., and C. Dosi (eds.), *The Economics of Nonpoint Pollution Control: Theory and Issues*. Dortrecht, The Netherlands: Kluwer Academic Press.

Shortle, J.S., and D.G. Abler. 1997. "Nonpoint Pollution," in Folmer, H., and T. Teitenberg (editors), *International Yearbook of Environmental and Natural Resource Economics.* Cheltenham, UK: Edward Elgar.

Shortle, J.S., and J. Dunn. 1986. "The Relative Efficiency of Agricultural Source Water Pollution Control Policies," *American Journal of Agricultural Economics* 68(3): 668-677.

Shortle, J.S., R.D. Horan, and D.G. Abler. 1998. "Research Issues in Nonpoint Pollution Control," *Environmental and Resource Economics* 11(3/4):571-585.

Shortle, J.S., R.D. Horan, M.O. Ribaudo, and D.G. Abler. 1998. "Point/Nonpoint and Nonpoint/Nonpoint Trading

Rules." Selected paper presented at the American Agricultural Economic Association annual meetings, Salt Lake City, UT, August 2-5, 1998.

Simons, P.C., H.A.J. Versteegh, A.W. Johgbloed, and P.A. Kemme. 1990. "Improvement of Phosphorus Availability by Microbial Phytase in Broilers and Pigs," *British Journal of Nutrition* 64:525-540.

Smith, M.E., and M.O. Ribaudo. 1998. "The New Safe Drinking Water Act: Implications for Agriculture," *Choices* (Third Quarter):26-30.

Smith, R.A., R.B. Alexander, and K.J. Lanfear. 1993. "Stream water quality in the conterminous United States-Status and trends of selected indicators during the 1980's," *National Water Summary 1990-91*. Water Supply Paper 2400. U.S. Dept. Interior, U.S. Geological Survey, pp. 111- 140.

Smith, R.A., R.B. Alexander, and M.G. Wolman. 1987. "Water-Quality Trends in the Nation's Rivers," *Science* 235:1607-1615.

Smith, R.A., G.E. Schwarz, and R.B. Alexander. 1994. *Regional Estimates of the Amount of U.S. Agricultural Land Located in Watersheds with Poor Water Quality.* Open-File Report 94-399. U.S. Dept. Interior, U.S. Geological Survey.

State of Florida. 1995. "Everglades improvement and management," *Florida Statutes*, Chapter 373.4592.

State of Maryland. 1984. *Maryland's Chesapeake Bay Program*. 1984 Annual Report.

Stavins, R.N. 1996. "Correlated Uncertainty and Policy Instrument Choice," *Journal of Environmental Economics and Management* 30:218-232.

Stephenson, K., W. Kerns, and L. Shabman. 1996. *Perspectives on Chesapeake Bay: Market-based Strategies for Chesapeake Bay Policy and Management.* Scientific and Technical Advisory Committee, Chesapeake Bay Program.

Sunding, D.L. 1996. "Measuring the Marginal Cost of Nonuniform Environmental Regulations," *American Journal of Agricultural Economics* 78(4):1098-1107.

Taylor, M.L., R.M. Adams, and S.F. Miller. 1992. "Farm-Level Response to Agricultural Effluent Control Strategies: The Case of the Willamette Valley," *Journal of Agricultural and Resource Economics* 17(1):173-185.

Thompson, L.C., J.D. Atwood, S.R. Johnson, and T. Robertson. 1989. "National implications of mandatory conservation compliance," *Journal of Soil and Water Conservation* 44(5):517-520.

Tomasi, T., K. Segerson, and J. Braden. 1994. "Issues in the Design of Incentive Schemes for Nonpoint Source Pollution Control," in Dosi, C., and T. Tomasi (eds.), *Nonpoint Source Pollution Regulation: Issues and Analysis*. Dordrecht, The Netherlands: Kluwer Academic Publishers.

Tsai, K., and J. Shortle. 1998. "Optimal Differentiation of Tax Rates and Base Definition in Nonpoint Pollution Control." Working Paper, Department of Agricultural Economics and Rural Sociology, Pennsylvania State University.

Tweeten, L. 1995. "The structure of agriculture: Implications for soil and water conservation," *Journal of Soil and Water Conservation* 50(4):347-351.

U.S. Congress, Congressional Budget Office. 1997. *Federalism and Environmental Protection: Case Studies for Drinking Water and Ground-Level Ozone*. Nov.

U.S. Congress, Office of Technology Assessment. 1990. *Beneath the Bottom Line: Agricultural Approaches to Reduce Agrichemical Contamination of Groundwater.* OTA-F-418. Nov.

______. 1995. *Targeting Enviornmental Priorities in Agriculture: Reforming Program Strategies*. OTA-ENV-640. September.

U.S. Department of Agriculture. 1993. *The USDA Water Quality Program Plan (1989).* Waterfax 000. Working Group on Water Quality, April.

U.S. Department of Agriculture, Animal and Plant Health Instpection Service. 1994. *Cryptosporidium and Giardia in Beef Calves.* National Animal Health Monitoring System report. Jan.

U.S. Department of Agriculture, Economic Research Service. 1994. *Agricultural Resources and Environmental Indicators.* Agricultural Handbook No. 705. Dec.

U.S. Department of Agriculture, Economic Research Service. 1997. *Agricultural Resources and Environmental Indicators.* Agricultural Handbook No. 712. July.

U.S. Department of Agriculture, National Agricultural Statistics Service, and Economic Research Service. 1990-93 Cropping Practice, Area Studies, Fruit, and Vegetable Survey data.

U.S. Department of Agriculture, National Agricultural Statistics Service, and Economic Research Service. 1996. *Agricultural Chemical Use: 1995 Field Crops Summary.* Ag Ch 1 (96). March.

U.S. Department of Agriculture, Natural Resources Conservation Service. 1996. *1994 Final Status Review Results* (unpublished).

U.S. Department of Agriculture, Natural Resources Conservation Service. 1998. *1996 Farm Bill Conservation Provisions: Environmental Quality Incentives Program*. USDA Fact Sheet. http://www.nhq.usda.gov/OPA/FB96OPA/eqipfact.htm.

U.S. Department of Commerce, and U.S. Environmental Protection Agency. 1993. *Coastal Nonpoint Pollution Control Program: Program Development and Approval Guidance.* Jan.

U.S. Environmental Protection Agency. 1990. *Rural Clean Water Program.* EPA 440/4-90-012. Office of Water, Sept.

_____ . 1991. "North Carolina Prepares to Implement Watershed/Basinwide Stream Pollution Control Strategy; TMDLs to be Established," *Nonpoint Source News Notes*, No. 13. June.

_____ . 1991b. *The Watershed Protection Approach: An Overview*. EPA/503/9-92/002. Office of Water, Dec.

_____ . 1992a. *Another Look: National Survey of Pesticides in Drinking Water Wells, Phase II Report.* EPA 579/09-91-020. Jan.

_____. 1992b. *National Water Quality Inventory: 1990 Report to Congress.* EPA 503-9-92- 006, April.

_____. 1993. *Guide to Federal Water Quality Programs and Information.* EPA-230-B-93-001. Office of Policy, Planning, and Evaluation, Feb.

_____. 1994a. *Drinking Water Regulations and Health Advisories.* Office of Water, May.

_____. 1994b. *The Quality of Our Nation's Water: 1992.* EPA841-S-94-002. Office of Water, March.

_____. 1995. *National Water Quality Inventory: 1994 Report to Congress*. EPA841-R-95- 005. Office of Water, Dec.

_____. 1997a. *Drinking Water Infrastructure Needs Survey First Report to Congress*. EPA 812-R-97-001. Jan.

_____. 1997b. "National Primary Drinking Water Regulations: Interim Enhanced Surface Water Treatment Rule Notice of Data Availability; Proposed Rule," *Federal Register*. Nov. 3:59486-59557.

_____. 1997c. "An Introduction to Water Quality Monitoring." http://www.epa.gov/OWOW/monitoring monintro.html.

_____. 1997d. "TMDL (Total Maximum Daily Load) Case Studies." http://www.epa.gov/OWOW/TMDL/docs.html.

_____. 1997e. *State Source Water Assessment and Protection Programs Guidance*. EPA 816- R-97-009. Office of Water, Aug.

_____. 1998a. *National Water Quality Inventory: 1996 Report to Congress*. EPA841-R-97-008. Office of Water, April.

_____. 1998b. *Drinking Water Contaminant List.* EPA 815-F-98-002. Office of Water, Feb.

U.S. Environmental Protection Agency, Office of Ground Water and Drinking Water. 1998. *Wellhead Program and State Ground Water Protection Programs (CSGWPP)*. http://www.epa.gov/OGWDW/csgwell.htm.

U.S. Environmental Protection Agency and U.S. Deptartment of Agriculture. 1998. *Clean Water Action Plan: Restoring and Protecting America's Waters.* Feb.

U.S. General Accounting Office. 1997. *Air Pollution: Overview and Issues on Emissions Allowance Trading Programs*. GAO/T-RCED-97-183. July.

U.S. Geological Survey. 1997. *Pesticides in Surface and Ground Water of the United States: Preliminary Results of the National Water Quality Assessment Program (NAWQA).* U.S. Dept. Interior, Aug.

Valigura, R.A., W.T. Luke, R.S. Artz, and B.B. Hicks. 1996. *Atmospheric Nutrient Input to Coastal Areas: Reducing the Uncertainties.* NOAA Coastal Ocean Program Decision Analysis Series No. 9. U.S. Department of Commerce, June.

Vandeman, A., J. Fernandez-Cornejo, S. Jans, and B. Lin. 1994. *Adoption of Integrated Pest Management in U.S. Agriculture*. AIB-707. U.S. Dept. Agr., Econ. Res. Serv., Sept.

Weaver, R.D. 1996. "Prosocial Behavior: Private Contributions to Agriculture's Impact on the Environment," *Land Economics* 72(2):231-247.

Weinberg, M., and J. E. Wilen. 1997. "A Closer Look at 'Inefficient' Policies: Options for Controlling Agricultural Nonpoint Source Pollution in California's San Joaquin River Basin." Selected paper presented at the American Agricultural Economics Association annual meeting, Toronto, Onatario, Canada, August.

Westenbarger, D., and D. Letson. 1995. "Livestock and Poultry Waste Control Costs," *Choices.* Second Quarter:27-30.

Weitzman, M. 1974. "Prices vs. Quantities," *Review of Economic Studies* 41:477-491.

Wetzstein, M.E., and T.J. Centner. 1992. "Regulating Agricultural Contamination of Groundwater Through Strict Liability and Negligence Legislation," *Journal of Environmental Economics and Management* 22:1-11.

Wischmeier, W.H., and D.D. Smith. 1978. *Predicting Rainfall Erosion Losses-A Guide to Conservation Planning.* AH-537. U.S. Dept. Agr., Science and Education Administration.

Wolf, S.A., and P.J. Nowak. 1996. "A regulatory approach to atrazine management: Evaluation of Wisconsin's groundwater protection strategy," *Journal of Soil and Water Conservation* 51(1):94-100.

Xepapadeas, A. 1991. "Environmental Policy Under Imperfect Information: Incentives and Moral Hazard," *Journal of Environmental Economics Management* 20:113-126.

Xepapadeas, A. 1992. "Environmental Policy Design and Dynamic Nonpoint Source Pollution," *Journal of Environmental Economics Management* 23:22-39.

Xepapadeas, A. 1994. "Controlling Environmental Externalities: Observability and Optimal Policy Rules", in Dosi, C., and T. Tomasi (eds.), *Nonpoint Source Pollution Regulation: Issues and Policy Analysis.* Dordrecht, The Netherlands: Kluwer Academic Publishers.

Young, D.L, D.J. Walker, and P.L. Kanjo. 1991. "Cost Effectiveness and Equity Aspects of Soil Conservation Programs in an Highly Erodible Region," *American Journal of Agricultural Economics* 73(4):1053-1062.