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The Economic Performance of Alternative Agricultural Nonpoint Pollution Controls

by

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THE ECONOMIC PERFORMANCE OF ALTERNATIVE AGRICULTURAL NONPOINT POLLUTION CONTROLS

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Concern for the environmental side effects of agricultural production among policy makers, regulatory authorities, and the general public has grown significantly in the U.S. and other developed countries in the last three decades. Environmental problems linked with agricultural production in the U.S. include nonpoint surface water and groundwater pollution by fertilizers, pesticides, animal manures, and sedimentation from eroded soils; loss of flora and fauna due to pesticides; and the conversion of wetlands to farm land. Of these problems, nonpoint source agricultural water pollution is widely recognized as one of the "last frontiers" of environmental regulation. There have been a variety of legislative and regulatory efforts in recent years at the federal and state levels directed at wetlands, while point sources of water pollution are covered under the federal Clean Water Act.

Whereas the federal government has taken the lead in point source control, states have traditionally had authority for dealing with nonpoint source water pollution. However, it would be fair to say that states have done little with their authority. Generally speaking, current state laws emphasize voluntary, education and information approaches, while the mandatory restrictions on production activities that do exist target so few farmers as to have little effect.[1] Farmers in all states are potentially liable for agricultural nonpoint water pollution under common law, while several states have legislation specifically addressing liability for groundwater contamination.[2]

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1. State laws adopted between 1972 and 1991 are reviewed briefly by Marc Ribaudo & Danette Woo, *Summary of State Water Quality Laws Affecting Agriculture*, *in* USDA, AGRICULTURAL RESOURCES: CROPLAND, WATER, AND CONSERVATION 50-54 (Economic Research Serv. AR-23, 1991); *see also* ORGANIZATION FOR ECONOMIC COOPERATION & DEV. (OECD), AGRICULTURAL AND ENVIRONMENTAL POLICY COORDINATION: RECENT PROGRESS AND NEW DIRECTIONS 53, 71 (1993). Useful discussions of recent experiences in specific states include: for Iowa, Rebecca S. Roberts & David R. Lighthall, *The Political Economy of Agriculture, Groundwater Quality Management, and Agricultural Research*, 27 WATER RESOURCES BULL. 437 (1991); for California, Jacques Franco et al., *California's Experience with a Voluntary Approach to Reducing Nitrate Contamination of Groundwater: The Fertilizer Research and Education Program (FREP)*, 49 J. SOIL & WATER CONSERVATION 76 (1994); and for Pennsylvania, Douglas B. Beegle & Les E. Lanyon, *Nutrient Management Legislation in Pennsylvania*, 49 J. SOIL & WATER CONSERVATION 84 (1994).

2. *See* Terence J. Centner, *Blameless Contamination: New State Legislation Regulating Liability for*

However, very few farmers to date appear to have been found liable for environmental damages.

The debate over Iowa's 1987 Ground Water Protection Act illustrates some of the unique political hurdles faced at the state level.[3] A proposal to tax pesticides was defeated in large part by opposition from state chemical dealers and state farmers, who were understandably wary of restrictions on their own activities that might put them at a competitive disadvantage relative to other states or countries.[4] At the federal level, concerns about putting one state at a disadvantage relative to others would be fewer, although concerns about international competitiveness would remain.[5]

The long-term outlook for more stringent action at the state level is unclear at this time. However, we would take it as axiomatic that any decision about whether or not to act, and the specific policy option to pursue if a state decides to act, should be based on the best available economic information about the merits of alternative policy options. The objective of this article is to evaluate the economic performance of alternative methods of controlling nonpoint source agricultural water pollution at the state level.

Section I reviews the nature and scope of U.S. nonpoint agricultural water pollution problems. Section II lays out major policy options, while section III develops some economic criteria for choosing among the options. Section IV reviews the evidence on the performance of the various options relative to the criteria. Section V concludes with some recommendations for policy makers at the state level.

I. Nonpoint Agricultural Water Pollution

A. Eutrophication

One of the leading surface water quality issues associated with agriculture in many areas of the U.S. is nutrient pollution by nitrogen and phosphorous.[6] Manufactured

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Agricultural Chemicals in Groundwater, 45 J. SOIL & WATER CONSERVATION 216 (1990); Theodore A. Feitshans, *Liability Issues in Groundwater Quality Protection*, 45 J. SOIL & WATER CONSERVATION 211 (1990).

3. *See* Roberts & Lighthall, *supra* note 1, at 439-41. Political factors influencing environmental policy in general at the state level are discussed by EVAN J. RINGQUIST, ENVIRONMENTAL PROTECTION AT THE STATE LEVEL: POLITICS AND PROGRESS IN CONTROLLING POLLUTION 80-103 (1993); SHERRY WISE & STANLEY R. JOHNSON, A COMPARATIVE ANALYSIS OF STATE REGULATIONS FOR USE OF AGRICULTURAL CHEMICALS, COMMODITY AND RESOURCE POLICIES IN AGRICULTURAL SYSTEMS 48-67 (1991).

4. Roberts & Lighthall, *supra* note 1, at 440.

5. This would be true provided the federal regulations treated producers in different states equally. If the regulations were designed so as to favor one state over another, such concerns would remain. Agricultural water quality regulations at the federal level might actually benefit farmers if the regulations, by reducing production, caused a sufficiently large increase in agricultural product prices. *See* David G. Abler & James S. Shortle, *Environmental and Farm Commodity Linkages in the U.S. and the E.C.*, 19 EUR. REV. AGRIC. ECON. 197, 209 (1992).

6. For more information on surface water pollution, see generally OECD, WATER POLLUTION BY FERTILIZERS AND PESTICIDES (1986); USDA, AGRIC. HANDBOOK NO. 705, AGRICULTURAL RESOURCES AND ENVIRONMENTAL INDICATORS (1994); C.M. Cooper, *Biological Effects of Agriculturally Derived Surface Water Pollutants on Aquatic Systems — A Review*, 22 J. ENVTL. QUALITY 402 (1993).

fertilizers contain nitrogen and phosphorous (and other nutrients) in varying degrees according to the type of fertilizer, while animal manure also contains these nutrients. Generally speaking, not all of the nutrients applied to fields or pasture are taken up by crops or grass. Nitrates (which are derived from nitrogen) are highly water soluble and bond only weakly with soil particles. Water running over or through the soil will remove some soluble nitrates from the soil, where they can leach into groundwater or run off into surface water. Although phosphates (derived from phosphorus) can also be leached from the soil, phosphorous is much less mobile in the soil than nitrogen. Phosphate losses from agricultural land occur primarily by means of soil erosion.

Nutrients that find their way into surface waters can lead to excess nutrient enrichment and eutrophication. An increase in nitrogen or phosphorous levels in slow-moving waters can stimulate algae growth, and the resulting effects on the aquatic ecology can be dramatic. As algae blooms and subsequently dies, it takes up dissolved oxygen, depleting the oxygen available for fish and other aquatic life. It can also block the sunlight needed by aquatic vegetation, causing the vegetation to die off. This loss in vegetation then moves up the food chain, leading to the death of fish and other aquatic life. Eutrophication of freshwater is usually due to phosphates, while nitrates are usually the cause of coastal water eutrophication.[7] Both nutrients tend to be important in the eutrophication of estuaries.

A number of recent assessments of water quality problems point to the fact that eutrophication is not a trivial issue. Among those river miles designated as impaired, agriculture was identified by the Environmental Protection Agency as a major or minor contributor in 72% of the cases; the corresponding figures for lakes and estuaries were 56% and 43%, respectively.[8] Agriculture tends to be the major cause of nutrient pollution in the Corn Belt and Central Plains, and is a large contributor in many other areas, including the Chesapeake Bay.[9]

Apart from reducing biodiversity, the loss of aquatic life from eutrophication can cause significant aesthetic and economic damages. The growth and subsequent decomposition of algae can be unsightly and generate foul odors, an obvious disamenity for those living or working near polluted waters. The productivity of commercial fisheries can be reduced, hurting those in the fishing industry and related industries. To the extent that this reduces supplies and thereby increases the price of fish and seafood, consumers are also harmed. Recreational fishing, boating, and swimming can also suffer, to the detriment of those who enjoy these activities and those who earn their living from them. To our knowledge no one has made a serious attempt to quantify the costs (in monetary terms) of eutrophication for the U.S. as a whole. However, it would be fair to say that the costs of eutrophication are at least as large as the costs of surface water damages from soil erosion, and as indicated below, estimates of annual costs of the latter run into the billions of dollars.

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7. WORLD RESOURCES INST., WORLD RESOURCES 1992-93, at 162-63, 184-85 (1992).

8. EPA, NATIONAL WATER QUALITY INVENTORY: 1992 REPORT TO CONGRESS 3-4 (1994). The percentages of all waters identified as partially or wholly impaired (regardless of cause) were 38% for rivers, 44% for lakes, and 32% for estuaries. *Id.*

9. USDA, *supra* note 6, at 60-61; WORLD RESOURCES INST., *supra* note 7, at 188-89.

B. Sedimentation and Turbidity

In a journey that may take many years, some of the soil blown or washed away by wind or water erosion moves from fields to waterways.[10] The remainder is redeposited on land somewhere else. Sediment suspended in water increases turbidity (i.e., it reduces water clarity). Turbidity and sedimentation can cause direct damage to fish and other aquatic life, while also destroying aquatic habitats. In still or slow moving waters, turbidity reduces the mixing of oxygen-rich surface water with deeper water. By blocking sunlight, turbidity also diminishes the production of oxygen in water through photosynthesis. Sediment can destroy fish eggs and increase the mortality of the eggs that do hatch. In addition, oysters and clams that must attach themselves to water bottoms free of heavy mud are at risk. There are a number of cases from the U.S. where aquatic species populations have been greatly diminished or destroyed by turbidity and sedimentation.[11]

For all these reasons, turbidity and sedimentation harm those who work in, and enjoy the benefits of, the commercial and sport fishing industries. Turbidity also reduces the aesthetic value of water for fishing, boating, and other uses, and it may also increase the likelihood of swimming and boating accidents. Sedimentation reduces the storage capacity of reservoirs and increases the costs of maintaining these facilities. It also fills bays, channels, and harbors. Sedimentation of rivers, other waterways, and even roadside ditches can increase the likelihood of flooding and the severity of flooding when it does occur. Sedimentation makes it more expensive to move water through irrigation systems and aqueducts. It also makes water treatment much more costly, particularly for drinking water.

Undoubtedly the most ambitious attempt to estimate the costs in monetary terms of soil erosion was made by Ribaudo[12]. Ribaudo's work has been criticized on a number of methodological grounds by Smith,[13] and indeed it is hard to take any particular cost figure too seriously. Be this as it may, Ribaudo's "best" estimate of the annual cost of soil erosion (in 1989 dollars) from all sources, including agriculture, is about $9 billion, with a range of estimates of about $5 billion to about $18 billion.[14] Ribaudo also estimates that agriculture accounted for about

10. *See* EDWIN H. CLARK ET AL., ERODING SOILS: THE OFF-FARM IMPACTS (1985).

11. Cooper, *supra* note 6, at 403-05.

12. *See* MARC O. RIBAUDO, WATER QUALITY BENEFITS FROM THE CONSERVATION RESERVE PROGRAM 4-22 (USDA Agricultural Economic Report No. 606, 1989). The literature in this area is reviewed by STEPHEN R. CRUTCHFIELD ET AL., THE BENEFITS OF PROTECTING RURAL WATER QUALITY: AN EMPIRICAL ANALYSIS 2-7 (USDA Agricultural Economic Report No. 701, 1995).

13. *See* V. Kerry Smith, *Environmental Costing for Agriculture: Will It be Standard Fare in the Farm Bill of 2000?*, 74 AM. J. AGRIC. ECON. 1076 (1992). Smith's principal criticism is that Ribaudo measured the cost to society associated with water treatment, dredging, and other activities designed to mitigate the damages of sedimentation by the costs of carrying out these activities. *Id.* at 1079. However, as Ribaudo himself notes, this underestimates the true cost because people generally prefer to prevent harmful activities such as sedimentation in the first place rather than deal with them after the fact. *See* RIBAUDO, *supra* note 12, at 13.

14. RIBAUDO, *supra* note 12, at 12.

53% of gross soil erosion in the U.S. in 1982.[15] Even if we use Ribaudo's lower estimate, it is clear that damages are significant.

C. Drinking Water Contamination

The third main water quality concern associated with agriculture is human health risks due to drinking water contamination by nitrates and pesticides.[16] One disease caused by ingestion of nitrates is methemoglobinemia, better known as blue-baby syndrome because bottle-fed infants less than six months old are particularly susceptible. This disease, which causes a reduction in the ability of blood to supply oxygen to the body, can lead to death. The incidence of this disease is unknown, but it is considered to be very rare. Nitrates are also suspected as a cause of cancer. They can react with other chemicals in the body to form N-nitrosamines, which are known to cause cancer in laboratory animals. However, there is no known relationship between human cancer and these compounds.

Like nutrients, there are a variety of possible fates for pesticides applied to fields and orchards. Pesticides can leach into groundwater or, when dissolved in runoff water or attached to eroded soil particles, wash into streams, rivers, lakes, and estuaries. Pesticides can also find their way into water resources via direct application to control aquatic weeds, wind drift, or overspray from aerial applications. The cleaning of application equipment or disposing of unused products into wells can pollute water resources as well.

The overall state of knowledge about the chronic effects of pesticides and pesticide residues on human health is quite limited, but concern has been raised about the consequences of low exposures over long periods of time. One cause for this concern is the fact that farmers and farm workers involved in the handling, mixing, and application of pesticides tend to have a higher incidence of lung cancer and other types of cancer.[17]

In recent years there have been several studies of the willingness of the general public to pay for drinking water free of nitrates, pesticides, and/or other agricultural

15. RIBAUDO, *supra* note 12, at 6. Agriculture's percentage of the costs of soil erosion would not necessarily be 53% of the figures above, but would instead depend on the percentage of erosion from agricultural sources that finds its way to water resources, the uses to which those resources are put, and other factors. *Id.* at 9-11.

16. For more information on health risks from nitrates, see KENNETH P. CANTOR ET AL., HEALTH EFFECTS OF AGRICHEMICALS IN GROUNDWATER: WHAT DO WE KNOW? AGRICULTURAL CHEMICALS AND GROUNDWATER PROTECTION: EMERGING MANAGEMENT AND POLICY (1987); SIDNEY S. MIRVISH, THE SIGNIFICANCE FOR HUMAN HEALTH OF NITRATE, NITRITE, AND N-NITROSO COMPOUNDS, NITRATE CONTAMINATION: EXPOSURE, CONSEQUENCE, AND CONTROL (1991). For more on groundwater pollution from nitrates and pesticides, see generally OECD, *supra* note 5; USDA, *supra* note 6; EPA, *supra* note 6; ELIZABETH G. NIELSEN & LINDA K. LEE, THE MAGNITUDE AND COSTS OF GROUNDWATER CONTAMINATION FROM AGRICULTURAL CHEMICALS (USDA Agric. Economic Report No. 576) (1987); Roy F. Spalding & Mary E. Exner, *Occurrence of Nitrate in Groundwater — A Review*, 22 J. ENVTL. QUALITY 392 (1993).

17. *See* CANTOR, *supra* note 16, at 31-37; Council on Scientific Affairs, *Cancer Risk of Pesticides in Agricultural Workers*, 260 JAMA 959, 960-63 (1988); WORLD HEALTH ORG., PUBLIC HEALTH IMPACT OF PESTICIDES USED IN AGRICULTURE 53 (1990).

pollutants.[18] These studies use the so-called "contingent valuation" method, in which people are asked how much (in monetary terms) cleaner water would be worth to them.[19] Estimates of the average willingness to pay per household, per year, for drinking water free of one or more agricultural pollutants range from less than $50 to more than $1,000. While figures in the neighborhood of $1,000 are simply too large to be credible, even $50 becomes significant when multiplied by thousands or millions of households.

II. The Policy Options

Environmental regulation in the U.S., like most environmental regulation worldwide, is characterized by a "command and control" mentality: producers are generally told not only what standards are required but also what must be done to meet those standards, in some cases specifying in great detail the actions to be taken to prevent or mitigate pollution.[20] The command and control approach is certainly one option to limit agricultural nonpoint water pollution, but even within this realm there are a wide variety of choices in terms of what to regulate and how to regulate it. This section lays out some of the policy options available at the state level.

A. The Nonpoint Nature of the Problem

Agricultural pollutants are nonpoint source pollutants, meaning they reach surface water and groundwater by diffuse and indirect pathways that are very difficult to predict or model in advance. The timing, frequency, and intensity of precipitation, which of course are unknown in advance, are critical in this regard. Soil characteristics (e.g., depth, density, and permeability) are also critical, and although these are measurable in principle they are often unknown in practice. In some cases, there may be long time lags (years or even decades) between polluting activities and the actual movement of pollutants into water resources.

The nonpoint character of agricultural water pollution places severe constraints on the options available to policy makers. One cannot control nonpoint pollution in the way, for example, that one can control the flow of sewage coming out of a pipe

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18. These studies are summarized in CRUTCHFIELD, *supra* note 12, at 2-7; Kevin J. Boyle et al., *What Do We Know About Groundwater Values? Preliminary Implications from a Meta Analysis of Contingent-Valuation Studies*, 76 AM. J. AGRIC. ECON. 1055, 1057-61 (1994).

19. Contingent valuation has been heavily criticized by some economists because the issues covered are often unfamiliar to people, because the time constraints inherent in any survey may limit the quality of responses, and because of the hypothetical nature of the questions. Since survey respondents have nothing to lose or gain, they may not think carefully about their answers or may not even tell the truth. However, proponents argue that contingent valuation can provide useful information provided that surveys are properly constructed and administered. *See* Peter A. Diamond & Jerry A. Hausman, *Contingent Valuation: Is Some Number Better than No Number?*, 8 J. ECON. PERSP. 45 (1994); W. Michael Hanemann, *Valuing the Environment Through Contingent Valuation*, 8 J. ECON. PERSP. 19 (1994); Paul R. Portney, *The Contingent Valuation Debate: Why Economists Should Care*, 8 J. ECON. PERSP. 3 (1994).

20. Some of the reasons for this are discussed in OECD, MANAGING THE ENVIRONMENT: THE ROLE OF ECONOMIC INSTRUMENTS 35-40 (1994) [hereinafter OECD, MANAGING THE ENVIRONMENT].

or pollutants coming out of a smokestack. The only options are to either (1) target farm-level decision variables that are known to be correlated to at least some degree with agricultural water pollution (e.g., the timing of manure applications), or (2) target estimates of the flows of nonpoint pollutants as opposed to the actual flows.[21]

Estimates of pollution flows could be quite crude or quite sophisticated. One long advocated proxy for targeting water quality problems caused by soil erosion, including phosphorous enrichment, is gross soil erosion as estimated by the Universal Soil Loss Equation (USLE) or modified versions of the USLE.[22] An analogous proxy that has been proposed for nitrate losses is the excess of manageable nitrogen applied over nitrogen removed by crops.[23] Estimates of excess nitrogen in practice tend to assume fixed nitrogen uptake coefficients for crops as well as fixed nitrogen delivery coefficients per pound of fertilizer and per animal in the form of manure. At the more sophisticated end of the spectrum, several field-level simulation models have been developed and widely used to estimate nutrient leaching and runoff, pesticide leaching and runoff, and erosion, using information on farm management practices, the weather, soil characteristics, and other relevant factors.[24]

B. Direct vs. Indirect Policies

An initial distinction among policy options would be between direct and indirect policies. Direct policies are those that give farmers a clear reason, of one sort or another, to change production practices so as to reduce pollution. These are policies that deal with the problem at the source, so to speak. An example would be a policy to reduce water pollution from manure through a limit on the number of livestock allowed per acre. Indirect policies, on the other hand, are those where the incentive to reduce pollution is not a clear and immediate result of government policy. Pollution may be reduced, but only through a chain of actions initiated by the policy that, in the end, induce farmers to voluntarily change production practices.

A good example of an indirect policy is research and development. For instance, government-funded R&D on alternatives to insecticides might lead to new crop varieties that are sufficiently pest-resistant to cut the use of insecticides, but this could take many years and would depend on farmers voluntarily adopting the new varieties, something that could hardly be guaranteed in advance. R&D has nevertheless attracted considerable interest because it appears, to many, to be a relatively "painless" way of cutting agricultural nonpoint pollution without forcing

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21. *See* James S. Shortle, *The Use of Estimated Pollution Flows in Agricultural Pollution Control Policy: Implications for Abatement and Policy Instruments*, 13 NE. J. AGRIC. & RESOURCE ECON. 277 (1984).

22. USDA, *supra* note 6, at 29.

23. *Id.* at 71.

24. *See* GAO, GROUNDWATER PROTECTION: VALIDITY AND FEASIBILITY OF EPA'S DIFFERENTIAL PROTECTION STRATEGY (1993) (GAO/PEMD-93-6); Donn G. DeCoursey, *Mathematical Models for Nonpoint Water Pollution Control*, 4 J. SOIL & WATER CONSERVATION 408 (1985).

farmers to do anything. Proponents of R&D note that both public and private agricultural research in the U.S. since World War II have been biased in favor of the development of chemical-intensive production techniques.[25] Thus, according to these proponents, we need a reorientation of the public agricultural research system if not also the private system.

Another popular indirect policy option is information and education programs. These programs provide farmers with information on practices for reducing nonpoint pollution, technical assistance in the adoption on these practices, or attempt to persuade farmers to adopt these practices. For instance, alternative production practices such as integrated pest management (IPM) and conservation tillage are currently important components of extension programs at state land grant institutions. Basic IPM practices, such as pest scouting to determine whether it would be profitable to apply pesticides, are now used by a significant fraction of farmers, although use of more sophisticated IPM techniques has been limited.[26] Conservation tillage is now being used by a significant fraction of grain producers, even on land classified as non-highly erodible and thus outside the purview of federal conservation compliance and "sodbuster" programs.[27]

C. Direct Regulation vs. Incentives

For economists, perhaps the most important distinction within the realm of direct policies is between direct regulation and incentive-based approaches. With direct regulation, farmers are told either what to do in order to reduce pollution or the cut in estimated pollution flows that is somehow expected of them. Failing to comply with the standards is a violation of the law, with all the civil or criminal penalties that this entails. For example, the law may prohibit the use of a certain pesticide. Incentives, on the other hand, either levy financial penalties on farmers for refusing to do what the law recommends or give farmers financial rewards for following the law's recommendations. Failing to follow the recommendations is not a violation of the law. If farmers voluntarily choose to pay the penalties or forgo the rewards, so be it. For example, a charge might be imposed on nitrogen fertilizer applications that exceed some per-acre threshold.

Direct regulation takes two basic forms: design standards and performance standards. Design standards tell producers specifically what to do in order to reduce pollution. Performance standards, on the other hand, give producers a pollution

25. Probably the best-known exposition of this point is NATIONAL RESEARCH COUNCIL, ALTERNATIVE AGRICULTURE (1989). Several studies support the contention that public and private research and development have favored chemical-intensive practices. *See, e.g.*, John M. Antle, *The Structure of U.S. Agricultural Technology, 1910-78*, 66 AM. J. AGRIC. ECON. 414 (1984); Wallace E. Huffman & Robert E. Evenson, *Supply and Demand Functions for Multiproduct U.S. Cash Grain Farms: Biases Caused by Research and Other Policies*, 71 AM. J. AGRIC. ECON. 761 (1989); Chris Fawson & C. Richard Shumway, *Endogenous Regional Agricultural Production Technologies*, 24 APPLIED ECON. 1263 (1992).

26. *See* ANN VANDEMAN ET AL., ADOPTION OF INTEGRATED PEST MANAGEMENT IN U.S. AGRICULTURE (USDA Agric. Information Bulletin No. 707) (1994).

27. USDA, *supra* note 6, at 125.

reduction target to achieve but leave it up to them as to how to achieve it. Examples of design standards would be restrictions on the timing, rate, and location of manure applications to the soil. A corresponding performance standard would be a limit on the estimated amount of nitrate runoff or leaching caused by the manure. Design standards are the essence of the command-and-control approach that is so common to environmental policy. This approach has brought several terms into the vocabulary of pollution control policy referring to prescribed techniques, including Best Practicable Technology (BPT), Best Available Control Technology (BACT), and, particularly in agriculture, Best Management Practices (BMPs).[28]

Incentive-based approaches, like direct regulation, can take two basic forms: design incentives and performance incentives. Following the example above, instead of imposing restrictions on manure application practices (design standard), we could give a subsidy to farmers who follow recommended practices or levy a charge on those who fail to do so (design incentive). Rather than mandating a limit on estimated nitrate runoff or leaching (performance standard), we could give a subsidy to farmers who fall within the limit or levy a charge on those who fail to do so (performance incentive).

Within the realm of design incentives or performance incentives, one can distinguish between subsidies and taxes (or charges). In terms of their impact on a farmer's production decisions, these two options can be viewed as different sides of the same coin. For example, suppose that current rate of nitrogen fertilizer application on corn, on a particular farm, is 175 pounds per acre (lb/ac), and that we wish to reduce this to 125. One option would be a tax of, say, $0.50 on every pound of fertilizer used above 125 lb/ac. Another option would be a subsidy of $0.50 on every pound of fertilizer below 175 lb/ac not used, down to the target of 125 lb/ac. In either case, the farmer gives up $0.50 for every pound of fertilizer used between 125 lb/ac and 175 lb/ac. The difference between the two options is of course that farmers gain with subsidies and lose with taxes, while the government loses money with subsidies and earns money with taxes.[29]

Performance standards and performance incentives could measure performance in a variety of ways. As noted above, one could use rough estimates of performance, such as the USLE or excess nitrogen, or one could use more complicated estimates derived from a field-level simulation model of nonpoint pollution flows.

D. Marketable Permits

The basic economic reason why nonpoint agricultural pollution is a problem to society at large is that it is an externality. Absent government policies of one type or another, the costs imposed on others never show up on a farmer's bottom line. In lieu of direct regulation or incentives, one option is to create a "market" in

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28. The environmental benefits and drawbacks of several BMPs are reviewed briefly by Terry J. Logan, *Agricultural Best Management Practices for Water Pollution Control: Current Issues*, 46 AGRIC., ECOSYSTEMS & ENV'T 223 (1993).

29. If farmers did not gain from the subsidies, they would forego them and continue with their previous production practices.

pollution through tradeable permits in estimated emissions or tradeable quotas in farm-level decision variables correlated with pollution (e.g., nitrogen fertilizer). Farmers would be required to have permits specifying their allowable pollution discharge levels or their allowable levels of input usage. The total supply of permits would be limited to some fixed value so that environmental quality goals would be achieved.[30] The initial allocation of permits among farms could be based on historical patterns of production, input usage, or estimated emissions.[31] However, unlike conventional permit systems, farmers would be able to trade permits among themselves as they so desired. Farmers who could limit emissions or input usage at a relatively low cost would have an incentive to use fewer than their allowed number of permits in order to be able to sell their surplus to other farmers who could not control pollution so easily.

Marketable permit schemes have, for the most part, been more theory than reality. However, in recent years the U.S. has begun experimenting with them to control air and water pollution, regulate the lead and oxygen in gasoline, and maintain wetlands.[32]

III. Economic Criteria for Policy Evaluation

In this section we review some important economic criteria in evaluating environmental policies for agriculture. We recognize that there are a variety of other criteria that can and should be used which are not discussed here, including federal and state constitutional considerations, distributional impacts (the distribution of benefits and costs of pollution control among various groups), political feasibility, and ethical considerations (e.g., environmental ethics).[33]

A. Cost-Effectiveness and Economic Efficiency

Environmental protection involves both benefits and costs. The benefits in this case derive from preventing eutrophication, sedimentation, or drinking water contamination. The costs in this case are the private and public sector activities

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30. A more complicated marketable permits scheme was developed by Marc J. Roberts & Michael Spence, *Effluent Charges and Licenses under Uncertainty*, 5. J. PUB. ECON. 193 (1976). This scheme was considered in the context of nonpoint source pollution by JAMES S. SHORTLE & DAVID G. ABLER, INCENTIVES FOR NONPOINT POLLUTION CONTROL, NONPOINT SOURCE POLLUTION REGULATION: ISSUES AND ANALYSIS (1994). The total number of permits is not fixed with this option. Rather, producers receive a subsidy for each one of their permits that they return to the regulatory agency and can also purchase additional rights from the agency.

31. Permits could also be allocated initially in some other manner so as to satisfy equity or political concerns. Alternatively, initial permit rights could be auctioned off to the highest bidder.

32. *See* OECD, MANAGING THE ENVIRONMENT, *supra* note 15.

33. Political and economic considerations are discussed in David G. Abler & James S. Shortle, *The Political Economy of Water Quality Protection from Agricultural Chemicals*, 20 NE. J. AGRIC. & RESOURCE ECON. 53 (1991). Legal, political, and economic factors are discussed in JAMES S. SHORTLE ET AL., THE POLITICAL ECONOMY OF GROUNDWATER PROTECTION, NON-POINT WATER QUALITY CONCERNS — LEGAL AND REGULATORY ASPECTS (1989). For the specific case of pesticides, *see* Erik Lichtenberg, *Alternative Approaches to Pesticide Regulation*, 21 NE. J. AGRIC. & RESOURCE ECON. 83 (1992).

which must be foregone because resources are being devoted to protecting water quality. Net benefits are the difference, in monetary terms, between benefits and costs.

Cost-effectiveness refers to the ability of a policy to achieve environmental protection goals at the lowest possible cost to society as a whole. One policy instrument is more cost-effective than another if it can achieve a given reduction in agricultural pollution at a lower cost. Economic efficiency refers to setting environmental protection goals so as to maximize net benefits to society as a whole. One policy is more efficient than another if it generates a higher net benefit. Economic efficiency is the more fundamental consideration because it addresses the choice of both the optimal level of environmental protection and the instruments to be used to achieve that protection, whereas cost-effectiveness takes the level of protection as a given. However, because quantitative information on the benefits of environmental protection is usually limited, as is the case for agriculture, cost-effectiveness can often be more easily determined.

A thorough economic analysis of the economic efficiency or cost-effectiveness of any policy option in a particular watershed would require an intensive study of the economic and biophysical systems involved.[34] Agricultural nonpoint water pollution is very site-specific: farming causes problems within some watersheds but not others, and within watersheds some farmers may generate more pollution than others. Because of this, the numbers coming out of any study would be specific to the watershed analyzed, making generalizations difficult. Thus, our approach here is to lay out some general factors that experience has shown are important determinants of economic efficiency and cost-effectiveness.[35]

1. Environmental Impact

One key consideration is the potential impact on the environment. All of the policy options listed above are obviously intended to have some impact. However, as discussed below, some might have so little impact that they fail to attain pollution reduction goals, others might improve some environmental problems while making others worse, while still others might actually be unambiguously bad for the environment.

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34. The steps involved in such an analysis would be: (1) estimating the impacts of the policy instruments on the production choices of farms and the corresponding costs to farms and others; (2) estimating the impacts of these behavioral changes on the proximate causes of environmental degradation; (3) estimating the impacts of changes in these proximate causes on environmental conditions; (4) estimating the impacts of changes in environmental conditions on "outcomes" such as public health, flora and fauna, etc.; and, for economic efficiency but not cost-effectiveness, (5) estimating the economic value of the changes in these outcomes.

35. A general discussion of these factors is found in Maureen L. Cropper & Wallace E. Oates, *Environmental Economics: A Survey*, 30 J. ECON. LITERATURE 675 (1992). For the specific case of nonpoint source pollution, *see* JOHN B. BRADEN & KATHLEEN SEGERSON, INFORMATION PROBLEMS IN THE DESIGN OF NONPOINT SOURCE POLLUTION POLICY, THEORY, MODELING AND EXPERIENCE IN THE MANAGEMENT OF NONPOINT SOURCE POLLUTION (1993).

In general, the environmental impact of a policy instrument depends on its effects on farm-level decision variables (e.g., nitrogen fertilizer application rates) that are correlated with environmental damages. Because of the site-specific nature of agricultural nonpoint pollution, the environmental impact also depends on the degree to which the policy targets farms causing pollution problems. Targeting in time can be important as well. For instance, the timing of manure applications can be at least as important as the volume of manure applications in affecting environmental losses of manure nutrients. Manure spread on frozen, snow-covered ground during the winter may be washed into streams with spring thaws, whereas application and incorporation during the growing season would permit crops to make use of manure nutrients.

In addition, the environmental impact of a policy depends on the magnitudes of the correlations between variables targeted by the policy option and environmental damages. Other things equal, a policy instrument will be more cost-effective and more efficient the stronger the association between the variables controlled and water pollution. It would be possible for a well-targeted policy, that has a strong effect on variables correlated with water pollution, to have little environmental impact if the magnitudes of the correlations were low. Similarly, it would be possible for a policy that targets variables highly correlated with water pollution to have little environmental impact if the policy had little effect on those variables.

2. Farm-Level Incentive Effects

Another critical factor in determining economic efficiency and cost-effectiveness is farm-level incentive effects. Does the policy give each farmer an incentive to control pollution in the way that is least expensive to society as a whole, or does it force or induce farmers to choose more expensive pollution control methods? Two policy options that yield the same reduction in pollution might have radically different costs to society as a whole. For example, a limit on livestock densities per acre of pasture and a limit on manure applications per acre of crop land could, in principle, each be structured to achieve the same estimated reduction in nitrate emissions. However, the economic impacts of these two options on livestock and crop production could be quite different, and there is no reason to assume that their social costs would be the same.

3. Inter-Farm Allocative Effects

Another critical factor in determining economic efficiency and cost-effectiveness is inter-farm allocative effects. Does the policy allocate the total reduction in pollution across farms in the least expensive manner to society as a whole? For example, suppose we wish to reduce the estimated leaching of some pesticide in a particular watershed by 50%. It does not logically follow that each and every farm in that watershed should reduce its estimated leaching by 50%. Some farms might be able, for any number of reasons, to cut their leaching much less expensively than others. In order to maximize the net benefits of pollution control to society as a whole, those producers who can clean up more cheaply should do more of the cleaning up. In fact, the least expensive option to society as a whole might actually involve putting some producers out of business entirely while letting others continue as they had before, without any changes in production practices.

The livestock densities versus manure applications example above illustrates how two policies designed to achieve the same reduction in estimated nitrate emissions would allocate the reduction across farms quite differently from each other. Livestock producers who did not have any crop land or who were not applying much manure to their crops would not be affected at all by the manure application restrictions but could be affected quite significantly by the density restrictions.

4. Administrative, Enforcement, and Compliance Costs

Effective environmental programs require institutional infrastructure and resources for design, implementation, evaluation, monitoring, and enforcement. These costs must therefore be considered when choosing between alternative approaches. This is particularly true in agriculture. Unlike many other industries, agriculture is not dominated by a few firms with a small number of plants. Other things equal, administration and enforcement are comparatively easy when the number of firms or plants to be monitored is small. Rather, agriculture is characterized by a large number of small, heterogenous, geographically dispersed producers. In general, administrative and enforcement costs depend on the amount of information required to implement (and as necessary, periodically revise) the instrument, the amount of information required to monitor compliance, incentives for noncompliance, the costs of sanctioning violators, and the administrative capability of regulatory agencies.

Compliance costs are also potentially significant. Some policy options could entail fairly onerous record-keeping and reporting requirements for farmers, whereas compliance with others could be fairly easy. In general, compliance costs depend on the amount of information required to comply with the law.

5. Innovation Effects

The final factor that we consider here is innovation effects. Does the policy stimulate the development of "environmentally friendly" production technologies, or do producers gain nothing by switching technologies?[36] Environmentally friendly technologies are those that permit producers to control pollution less expensively than current technologies. R&D is the only policy option listed above that specifically seeks to develop alternative technologies. However, other policies might indirectly encourage more environmentally friendly R&D in the private and public sectors.

For example, a tax on pesticides would encourage producers to use fewer pesticides. This could be accomplished to some extent by taking greater advantage

36. The general process of technology responding to economic incentives created by public policy or other forces is known as induced innovation. For discussions of the potential for environmental policy to stimulate the development of new technologies, *see* Susse Georg et al., *Clean Technology — Innovation and Environmental Regulation*, 2 ENVTL. & RESOURCE ECON. 533 (1992); René Kemp et al., *Supply and Demand Factors of Cleaner Technologies: Some Empirical Evidence*, 2 ENVTL. & RESOURCE ECON. 615 (1992).

of off-the-shelf alternatives like IPM. However, private seed companies would probably still see the pesticide tax as enlarging the market for crop varieties with more built-in pest resistance, and as such would increase their R&D in this area. Similarly, private firms in the business of supplying the beneficial insects used as part of many IPM programs would be encouraged to do even more work in selectively breeding insects with desirable properties. Whether researchers in the public sector would also be encouraged to do more environmentally friendly R&D is hard to say. Unlike the private sector, the public sector rarely has a clear bottom line on which to base decisions. In any event, about 70% of U.S. agricultural research is conducted by the private sector.[37]

B. Environmental Risk

Policy options are likely to vary with respect to their ability to offer reliable environmental protection. Even a policy that performs well with respect to environmental protection goals, on average (good environmental impact), might do very well some of the time and very poorly at other times (high environmental risk). For example, one could be reasonably sure that farms would comply with a restriction on the use of a specific pesticide, provided it were adequately enforced. On the other hand, farm-level responses to economic instruments such as taxes and subsidies could vary across space and time. For example, the degree to which a tax on a specific pesticide reduced pesticide usage would depend on the total cost of the pesticide, including the tax, relative to what farmers could expect to gain if they went ahead and used the pesticide anyway. The latter, in turn, would be a function of the anticipated effectiveness of the pesticide and the anticipated prices of the agricultural products on which the pesticide was used, both of which could vary significantly from one year to another.

Environmental policy in the U.S. suggests a strong "revealed preference" for certainty in environmental protection. For example, EPA pesticide regulations designed to protect worker safety are based on very cautious estimates of the potency of pesticides, the number of years of exposure among those who work with pesticides, and the intensity of exposure per year.[38] However, the degree to which this apparent aversion to risk is due to distortions in regulatory and legislative processes, a failure to adequately educate the public about risks, or a genuine public dislike of risk is the subject of much debate.[39]

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37. *See* WALLACE E. HUFFMAN & ROBERT E. EVENSON, SCIENCE FOR AGRICULTURE: A LONG-TERM PERSPECTIVE 96 (1993).

38. Carolyn R. Harper & David Zilberman, *Pesticides and Worker Safety*, 74 AM. J. AGRIC. ECON. 68, 73-74 (1992); *see also* Maureen L. Cropper et al., *The Determinants of Pesticide Regulation: A Statistical Analysis of EPA Decisionmaking*, 100 J. POL. ECON. 175 (1992).

39. Some of the diverse points of view on this issue can be found in COMM. ON SCIENCE, SPACE, AND TECHNOLOGY, U.S. HOUSE OF REPRESENTATIVES, STRENGTHENING RISK ASSESSMENT WITHIN EPA (1994). *See, e.g.*, *id.* at 10-18 (viewpoint of Sally Katzen); *id.* at 59-80 (viewpoint of Ellen K. Silbergeld); *id.* at 102-19 (viewpoint of Chemical Manufacturers Association).

IV. Economic Performance of the Policy Options

A. Research and Development

The environmental impact of R&D as an agricultural pollution control policy depends on whether and when researchers are able to develop profitable alternatives to agricultural chemicals in crop production and management practices associated with pollution. It is generally accepted that R&D has not yielded any practical, large-scale substitutes for agricultural chemicals. There are many promising possibilities, particularly involving genetic engineering, but development will probably require many years.[40]

The environmental impact of R&D also depends on whether alternatives to agricultural chemicals would actually reduce agricultural chemical usage. Suppose a new variety of corn were developed with a built-in toxin to corn rootworm, reducing the need for insecticides to achieve any given level of control of this pest. Also suppose that the new variety, taking into account the reduced insecticide usage per acre, were less costly to producers than current varieties, so that it were widely adopted. At the market level, competition among corn producers would pass the reduction in the cost of production along to consumers in the form of lower corn prices, which would stimulate the demand for corn. If this stimulus were great enough, the result could be perverse. Producers might actually increase insecticide use in order to all the more effectively control rootworm and thus all the more effectively meet the expanded demand for corn. Previous research indicates that alternative production technologies could actually work to increase fertilizer and pesticide usage through mechanisms such as this.[41] Moreover, returning to the corn rootworm example, producers might have an incentive to switch lands from other crops to corn, which is significant because corn is among the most chemical-intensive of all crops.[42]

B. Information and Education

The potential environmental impact of information and education, like R&D, must be rated as limited. Without good alternatives, significant changes in existing farm management practices would offer only small or even negative economic gains to farmers.[43] One could argue that farmers should change production practices in

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40. *See, e.g.*, WORLD BANK, TECHNICAL PAPER NO. 133, AGRICULTURAL BIOTECHNOLOGY: THE NEXT "GREEN REVOLUTION"? 23 (1991). Existing alternatives to agricultural chemicals are discussed in NATIONAL RESEARCH COUNCIL, *supra* note 25, at 135-64.

41. David G. Abler & James S. Shortle, *Technology as an Agricultural Pollution Control Policy*, 77 AM. J. AGRIC. ECON. 20, 26-28 (1995).

42. *See* USDA, *supra* note 6, at 83-85, 95-98. In general, shifting acreage from one crop to another might be good or bad for the environment depending on fertilizer usage, pesticide usage, and management practices for each crop.

43. *See* Glenn Fox et al., *Comparative Economics of Alternative Agricultural Production Systems: A Review*, 20 NE. J. AGRIC. & RESOURCE ECON. 124 (1991); Linda K. Lee, *A Perspective on the Economic Impacts of Reducing Agricultural Chemical Use*, 7 AM. J. ALTERNATIVE AGRIC. 82 (1992).

order to protect their own water supplies from contamination (at least those farmers with private, on-farm wells) or protect their own health from pesticides. Here too, however, the perceived benefits to farmers of significant changes in practices would be small.[44] The conclusion is that information and education cannot be relied on to protect water quality. They can help, but real improvements in water quality will require regulatory measures or economic incentives.

C. Direct Regulation

1. Design Standards

The environmental impact of design standards depends in large measure on the farm-level variables targeted by the standards and the particular site subject to the standards. Some variables might be highly correlated with water pollution on certain farms and at certain times but not correlated, at all, on other farms or at other times. For example, simulation analyses indicate that restrictions on nitrogen timing and application rates targeted toward intensively managed farms with surplus nitrogen can significantly cut leaching and runoff at a relatively low cost, although broader restrictions on nitrogen use applied to an entire watershed would do little to reduce pollution and would be significantly more costly.[45]

Design standards in practice often involve mandated use of particular Best Management Practices (BMPs). However, the environmental impacts of some BMPs are not unambiguously positive.[46] For example, conservation tillage, contouring, filter strips, and strip cropping, all tend to reduce erosion and the volume of runoff at the expense of additional leaching and higher pollutant concentrations in the runoff that does occur. This reflects the simple fact that excess nutrients have to go somewhere; if not into surface waters then perhaps into groundwater.[47] Whether or not this shift in pollution from one outlet to another is viewed as acceptable depends on the relative social costs in a particular watershed of erosion and runoff versus leaching. Regulators mandating one or more of these four BMPs but still seeking to reduce leaching would have to mandate other BMPs or implement some other pollution control policy.

The performance of different policy options in regard to farm-level incentive effects depends on the information available to farmers as opposed to regulators. If regulators had better information than farmers about the costs and efficacy of changes in farm-level variables needed to meet environmental protection goals, then design standards would be the preferred choice in giving farms proper incentives to

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44. *See* E. Douglas Beach & Gerald A. Carlson, *A Hedonic Analysis of Herbicides: Do User Safety and Water Quality Matter?*, 75 AM. J. AGRIC. ECON. 612 (1993).

45. *See, e.g.*, Scott L. Johnson et al., *The On-Farm Costs of Reducing Ground Water Pollution*, 73 AM. J. AGRIC. ECON. 1063 (1991); Harry P. Mapp et al., *Economic and Environmental Impacts of Limiting Nitrogen Use to Protect Water Quality: A Stochastic Regional Analysis*, 76 AM. J. AGRIC. ECON. 889 (1994); Michael L. Taylor et al., *Farm-Level Response to Agricultural Effluent Control Strategies: The Case of the Willamette Valley*, 17 J. AGRIC. & RESOURCE ECON. 173 (1992).

46. *See* Logan, *supra* note 28, at 228.

47. Nitrogen can also be lost to the atmosphere through processes known as denitrification and volatilization.

control pollution. However, farms are very heterogenous, and it is unrealistic to expect a regulatory agency to know more about the soils, weather, pests, etc. on a particular farm than the operator of that farm. Regulatory agencies are likely, in fact, to have very poor information about individual farms. Design standards prevent farmers from using their specialized information to figure out cheaper ways of achieving pollution reduction goals than those mandated by the standards. They therefore rate poorly in terms of farm-level incentive effects.[48] For similar reasons, design standards also rate poorly in regard to inter-farm allocative effects. Regulators with little information cannot be expected to set standards that correspond to the relative costs of pollution control across farms.

Design standards perform better than most other regulatory and incentive options in terms of administration and enforcement costs. There is no need to monitor or estimate pollution flows at the farm level, as would be necessary with performance standards or performance incentives. Nor is there any need to determine the level of incentives needed to achieve the desired results. On the other hand, they perform worse than most other options with respect to innovation effects. They would tend to encourage innovation to enable farmers to more profitably comply with the standards, but innovations directed at methods of pollution control, not on the list of approved or required methods, would be discouraged.

2. *Performance Standards*

Performance standards are preferable to design standards on a number of grounds. They are ranked highly with respect to farm-level incentive effects because they give farmers an inducement to use their specialized information to minimize costs while still attaining pollution reduction goals. Performance standards are also ranked well in regard to innovation effects because they give farmers the ability to adopt lower-cost methods of pollution control, thereby stimulating the demand for R&D to develop such methods. Performance standards score poorly on inter-farm allocative effects because regulators lack the information necessary to more strictly target the farms that can control pollution at lower costs. However, performance standards are no worse than design standards in this regard.

Performance standards excel at minimizing environmental risk. Provided that performance is measured accurately and properly enforced, regulators can be certain that water quality protection goals will be met, leaving aside unforseen variations due to the weather and other random factors. Design standards can approach this level of certainty provided that the variables targeted by the standards are closely correlated with pollution flows.

The Achilles heel of performance standards is the accuracy and cost of estimates of nonpoint water pollution flows. Accuracy is essential if this approach is to have a positive environmental impact, withstand legal challenge, and gain policy legitimacy. Crude proxies such as the USLE have obvious limitations.[49] At the

48. Simulation results in this regard may be found in William T McSweeny & James S. Shortle, *Probabilistic Cost Effectiveness in Agricultural Nonpoint Pollution Control*, 22 S. J. AGRIC. ECON. 95, 100-01 (1990).

49. *See, e.g.*, STEFANO PAGIOLA, COST-BENEFIT ANALYSIS OF SOIL CONSERVATION, ECONOMIC AND

same time, more sophisticated field-level simulation models would require an enormous amount of information and computing resources to be applied to the hundreds or thousands of farms in a typical watershed. If the regulatory agency collected this information and supplied the computing resources, administrative costs would increase; if some of the burden were shifted on to farmers (e.g., to supply information about their own soils), compliance costs would increase. Even the best models have not yet reached a point where they can be used regularly to provide accurate estimates of pollution losses.[50]

D. Economic Incentives

Like design standards, the environmental impact of design incentives depends on the farm-level variables targeted by the incentives. Some variables might be highly correlated with pollution flows in particular locations and at particular times but only weakly correlated at other locations or times. Unlike design standards, however, the environmental impact of design incentives also depends on the effects of the incentives on farm-level decision making. For example, the potential economic impacts of fertilizer taxes have been the subject of a number of studies.[51] These studies demonstrate that the impacts of fertilizer taxes would depend on a variety of hard-to-predict factors, especially the price responsiveness of fertilizer demand. The literature generally suggests that fertilizer demand is not highly responsive to fertilizer prices, at least in the short run. If this were true, high tax rates would be needed to significantly reduce fertilizer use.

Performance incentives suffer from similar problems of control over farm-level decision variables. For example, the environmental impact of a tax on excess nitrogen per acre (nitrogen from fertilizer and manure left over after uptake by crops) would depend on a whole host of economic factors, including the price responsiveness of fertilizer demand, the responsiveness of the stock of farm animals to the tax on manure, and the price responsiveness of the supply of acreage to crop and livestock production (because the tax would tend to encourage farmers to spread production over more acres).

The effects of design incentives on farm-level incentives to control pollution and the allocation of pollution reduction across farms depend on the degree to which the variables targeted by the incentives are correlated with pollution flows. For example, fertilizer taxes cannot be designed to discriminate well between polluted and nonpolluted watersheds, or between polluting and nonpolluting farms within

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INSTITUTIONAL ANALYSES OF SOIL CONSERVATION PROJECTS IN CENTRAL AMERICA AND THE CARIBBEAN 28-29 (World Bank Environment Paper No. 8, 1994).

50. *See* NATIONAL RESEARCH COUNCIL, SOIL AND WATER QUALITY: AN AGENDA FOR AGRICULTURE (1993).

51. *See, e.g.*, Alison Burrell, *The Demand for Fertilizer in the United Kingdom*, 40 J. AGRIC. ECON. 1 (1989); Mark Denbaly & Harry Vroomen, *Dynamic Fertilizer Nutrient Demands for Corn: A Cointegrated and Error-Correcting System*, 75 AM. J. AGRIC. ECON. 203 (1993); Wen-yuan Huang & Michael Le Blanc, *Market-Based Incentives for Addressing Non-Point Water Quality Problems: A Residual Nitrogen Tax Approach*, 16 REV. AGRIC. ECON. 427 (1994); Taylor et al., *supra* note 45, at 180-81.

watersheds. Absent significant expenditures on enforcement, any attempt to discriminate in this regard could be easily evaded, particularly at the state level where the tax could only be enforced at within-state fertilizer supply points. Performance incentives do better on this score. For example, in the case of nitrogen losses, excess nitrogen is more highly correlated with runoff and leaching than fertilizer applications.[52] A tax on purchased nitrogen fertilizer would encourage producers to substitute other sources of nitrogen for purchased fertilizer, partially defeating the objective of the tax, whereas a tax on excess nitrogen would make such a substitution pointless.

Performance incentives do better than design incentives regarding farm-level incentive effects and inter-farm allocative effects. A number of studies have demonstrated that incentives applied directly to estimated soil loss can reduce soil loss at less cost to farmers than incentives applied to particular erosion control practices.[53] Similarly, several recent studies have indicated that excess nitrogen can be reduced at lower cost to farmers by the application of incentives directly to excess nitrogen than to fertilizer or other nutrient management practices.[54]

In a static economic setting, the costs of administering and enforcing incentives would not differ too much from their counterpart standards. These costs would range from moderate to large in the cases of design incentives and standards, and moderate to prohibitive in the cases of performance incentives and standards. In a dynamic economic setting, however, the levels of incentives might need to be adjusted periodically to maintain their effectiveness. For example, the long-term trend in real (inflation-adjusted) prices of farm products is downward; real prices are significantly lower now than they were fifty years ago.[55] This means, for example, that a tax on excess nitrogen set today at a level sufficient to achieve water quality goals at a reasonable cost to farmers might subsequently become very high relative to crop and livestock prices.

One important advantage of performance incentives, in general, is that they have good innovation effects. Because producers are taxed or subsidized on the basis of their environmental performance, they have a clear incentive to adopt new technologies to improve their performance. This incentive, in turn, gives rise to incentives to develop new pollution prevention or abatement technologies. In the case of nonpoint pollution, however, some of the advantages of performance incentives are lost because of the need to use estimated performance measures rather than actual performance measures.

For reasons noted above, incentives entail a high environmental risk relative to other policy options. The incentives offered may or may not be sufficient in any one place or at any one point in time to induce the desired changes in farmers'

52. NATIONAL RESEARCH COUNCIL, *supra* note 50, at 266.

53. *See, e.g.*, McSweeny & Shortle, *supra* note 48, at 100-01.

54. *See, e.g.*, Huang & Le Blanc, *supra* note 51, at 435-36.

55. Compare the producer price index for farm products with other producer price indices in COUNCIL OF ECONOMIC ADVISORS, STATISTICAL TABLES RELATING TO INCOME, EMPLOYMENT, AND PRODUCTION, ECONOMIC REPORT OF THE PRESIDENT 347-51 (1995).

behavior. Insufficient incentives would fail to meet pollution reduction objectives, while overly strong incentives would reduce pollution by more than the economically efficient amount.

E. Marketable Permits

Marketable permits could be issued for inputs correlated with pollution or for emissions as estimated by formulas such as those mentioned above. In either case, as scarce "commodity" permits would acquire a price, with farmers wishing to augment their permit holdings purchasing from farmers willing to sell their holdings. The fact that a permit could only be bought at a price would raise the cost to the farm of inputs or estimated emissions. Even farms using no more than their initial allocation of permits would always face an opportunity cost because they could have sold those permits to someone else.

Provided that permits were issued for estimated emissions and provided that estimated emissions were closely correlated with actual emissions, they would do very well with respect to environmental impact and environmental risk. Since the total number of permits is always fixed, the regulatory agency can restrict the number to achieve whatever environmental protection goal is desired. Moreover, given that the permit system were adequately enforced, the agency could be sure that this goal would be met, apart from variability in actual emissions owing to the weather and other random factors.

Under these conditions, permits also do well regarding farm-level incentive, innovation, and inter-farm allocative effects. Since (estimated) pollution would have a market price, farms would have strong incentives to control pollution, including the use of new pollution prevention or abatement technologies. Moreover, the regulatory agency would not need to know anything about the costs of controlling pollution on one farm versus another in order to optimally allocate the reduction in pollution across farms. Through permit trading, producers could sort this matter out for themselves. The beauty of a price system is that the market price conveys information about supply and demand conditions, saving everyone the cost of collecting this information separately. In this case, the price of a permit signals what farms with high costs of pollution control are willing to pay and what farms with low costs of control are willing to accept.

Administration, enforcement, and compliance costs would probably not be too much of a burden. Farmers are quite familiar with participating in auctions and other competitive forums. Owing to federal farm commodity programs, they are also quite familiar with external constraints on resource use in the form of output quotas and acreage restrictions. Similarly, agricultural agencies, at least at the federal level, have considerable experience in administering such programs.

The advantages of a marketable permit system could break down if the market for permits were "thin." Suppose that the farms in some watershed were relatively similar to each other in terms of the costs of controlling emissions. In this case, farms would have little to gain from trading permits. The allocation of emission permits across farms in this setting would closely correspond to the initial allocation of permits, so that we would basically find ourselves back in the case of performance standards (the standards being the initial permit allocations). The advantages of marketable permits are only realized to the extent that farms are different from each other.

V. Conclusions

It should be evident that there is no "one size fits all" solution to controlling nonpoint source agricultural pollution. However, policies that target estimated performance are probably better than policies that target design, provided that performance is estimated in a relatively simple yet reliable manner (e.g., excess nitrogen). More complicated methods of estimation involving simulation models are not practical for policy making purposes at this time. Within the class of performance-based instruments, marketable permits appear to be superior to performance incentives on most of the policy evaluation criteria. Permits are also superior to performance standards, provided that the farms subject to regulation differ enough from each other in order to make an active permit market. If farms are relatively homogenous, there is little point in trying to set up a permit market, and performance standards might as well be used.