Agriculture and the Environment

by

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Abstract: The distinctive nature of environmental quality problems in agriculture—an industry based on the extraction of highly variable natural resources under stochastic conditions—has important implications for policy design. First, we examine the source of environmental quality problems and the strength of incentives for resource stewardship that may incidentally induce farmers to protect environmental quality. In turn, we examine environmental policy design under two features that are pervasive in agriculture: (1) heterogeneity caused by resource variability and (2) uncertainty. Next, we examine the effects of interactions between agricultural, environmental, and resource policies. Finally, we review important areas for further research.

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Outline

AGRICULTURE AND THE ENVIRONMENT

1. Introduction

Agriculture provides a wide variety of environmental amenities and disamenities. On the positive side, farms provide open space and scenery. On the negative side, agriculture is a major contributor to numerous environmental problems. Nitrate and pesticide runoff impair drinking water quality and degrade habitat for aquatic organisms including fish, affecting commercial fisheries and recreational uses of estuaries, lakes, and streams. Bacterial contamination from animal wastes impairs drinking water quality and contaminates shellfish. Odor from concentrated livestock facilities worsens the quality of life in nearby residential areas. Erosion-induced sedimentation of waterways increases drinking water treatment costs and accelerates the need for dredging to maintain navigability. Pesticide exposure causes both acute and chronic illness among farmers and farm workers, while pesticide residues on foods may also threaten human health. Ecological damage from agriculture includes kills of fish, birds, animals, and invertebrates from pesticides and, most important, habitat loss from conversion of wetlands and grasslands. Heavy metals like selenium and arsenic in drainage water have been implicated in wildlife kills and reproductive problems and can pose hazards to human health. Negative externalities also occur within agriculture. Salinization of rivers by irrigation runoff damages crop production in downstream areas. Upslope irrigation may cause drainage problems in downslope areas. Pesticide drift kills bees and thus impairs orchard pollination.

These problems have spawned a large, wide-ranging literature exploring efficient and equitable policy design. This chapter reviews some of the major developments in that literature, concentrating on the portion that addresses the major features distinguishing agriculture from other industries.

We begin by considering the size, scope, and origins of environmental problems in agriculture. Agriculture involves extraction of renewable resources under naturally occurring conditions. Agricultural productivity has traditionally depended on the natural resource base of agriculture, giving farmers economic incentives for conserving that resource base. Protection of environmental quality has historically been a side effect of those conservation efforts. These economic incentives for resource conservation have traditionally been referred to as "stewardship." Section 2 explores the implications of stewardship for environmental policy.

The resource base of agriculture varies substantially both across growing regions and across farms and fields within growing regions. As a result, both agricultural productivity and environmental quality exhibit significant heterogeneity. Section 3 explores the implications of heterogeneity for environmental policy design.

As noted above, agricultural production occurs largely under naturally occurring conditions. Thus, stochastic factors exert significant influence on both agricultural productivity and environmental quality. Section 4 explores the implications of uncertainty for environmental policy design.

For millennia, agriculture has been central to human existence. The irreducible human need for food has led governments in virtually all countries to adopt policies relating specifically to the agricultural sector. Environmental policies aimed at agriculture operate in combination with these farm-sector policies. Section 5 explores interactions between environmental and agricultural policies.

Finally, Section 6 sums up the major lessons of this literature and highlights topics of special interest for further research.

2. Agriculture, Stewardship, and the Environment

Farming is, at bottom, a resource extraction industry. Both crop and livestock production involve harvesting biota, that is, renewable natural resources produced by biological processes. Both utilize as intermediate inputs a variety of natural resources, such as soils, water, genetic material, non-crop plant life, and naturally occurring fauna that mitigate damage caused by pest species. These natural resources may simultaneously influence environmental spillovers from agricultural production like water pollution, pesticide poisonings, or scenic amenities.

The farm environment has also traditionally been an important source of direct consumption goods for farmers and their families. Hunting and fishing have historically provided significant shares of farmers' diets (and continue to do so in many developing countries). Groundwater and local streams can be major sources of drinking water. Protecting wildlife habitat, water quality, and other aspects of environmental quality at the local level can thus be equivalent to protecting farmers' standard of living.

Until recently, agriculture was considered a clean industry, largely because farmers' well-being depended on the resource base of agriculture and on local environmental quality. Farmers were thought to be stewards of both in their own self-interest. Even today, stewardship is often invoked as a solution to environmental problems in agriculture, and attempts to popularize more environment-friendly farming practices remain the major form of environmental policy in agriculture, at least in developed countries.

In this section, we consider stewardship as a means of dealing with environmental problems. We begin by developing a formal model of the market failures underlying these environmental problems. We then use the model to formalize the notion of stewardship. Finally, we consider the empirical evidence on the strength of these stewardship incentives and discuss the relationship between stewardship and technical change. The section as a whole addresses some of the central questions arising in discussions of sustainable agriculture, e.g., the extent to which farmers have incentives for protecting environmental quality voluntarily and how those incentives are influenced by technical change.

2.1 Agriculture and Environment: Theory

We begin with two formal models for characterizing the market failures underlying environmental problems in agriculture. The first is an output-oriented model in which the natural resource base enters only implicitly. The second is an input-oriented model that can incorporate those resources explicitly.

Overall, environmental spillovers are perhaps best conceptualized as arising from a multiple-output production system in which agricultural production and environmental quality are produced simultaneously from a given vector of inputs. Let y be a vector of agricultural products, q be a vector of environmental impacts, and x be a vector of inputs. Agricultural technology is then a set $T = \{(y,q,x): x \text{ can produce } (y,q)\}$.

The output-oriented model represents this technology using a joint cost function for agricultural output and environmental quality $C(y,q,w) = \min\{wx: (y,q,x) \in T\}$, where w is a vector of input prices. Let $U(y,q)$ represent society's gross benefit from the consumption of agricultural output and environmental quality. We assume that these benefits are strongly separable from other forms of consumption or, equivalently, that the agricultural sector accounts for a share of the overall economy sufficiently small that income effects are negligible. Optimal joint production of agricultural products and environmental quality is found by choosing (y,q) to maximize net benefits $U(y,q) - C(y,q,w)$. The necessary conditions are

$$U_y(y,q) - C_y(y,q,w) = 0$$

$$U_q(y,q) - C_q(y,q,w) = 0,$$

i.e., the marginal benefits of agricultural output and environmental quality should be equated to their marginal costs. (Subscripts denote partial derivatives.)

If market (inverse) demands for agricultural output and environmental quality equal $U_y(y,q)$ and $U_q(y,q)$, respectively, then perfectly competitive markets will generate the socially efficient levels of agricultural output and environmental quality in equilibrium. But in most cases, environmental quality problems arise in agriculture because effective market demand for environmental quality in agriculture is incomplete or lacking. For example, farmers are not required to pay for disposal of sediment, nutrients, or pesticides into surface or ground waters. Similarly, farmers cannot charge for the open space, greenery, and scenic views their farms provide for neighboring residents and for passersby. Both the environmental resources farmers provide and those they use are thus subject to open access exploitation. At bottom, the lack of markets for these environmental resources is due to the extreme difficulty—or even downright impossibility—of establishing and enforcing clear property rights. (For a more complete discussion of property rights issues see Bromley 1991.) Most scenic amenities from agriculture, for example, are public goods. The consumption of farm scenery is nonexclusive and non-rival. Moreover, since the marginal cost of providing scenery to an additional consumer is minimal, leaving these scenic amenities unpriced can be efficient at the margin. But leaving such amenities unpriced fails to provide farmers with incentives to forego development of their land, even when doing so is in the public interest. Open access occurs in water pollution from agricultural emissions of nutrients, sediment, or pesticides for a somewhat different reason. Emissions of agricultural pollutants are diffuse and enter water bodies at numerous different points. Furthermore, it is difficult to distinguish agricultural emissions of nutrients and sediments from those that occur naturally. It is impossible to restrict access to these water bodies, and hence impossible to levy charges for access to them.

In the absence of markets for environmental quality, farmers tend to treat it as freely disposable. In an unregulated market, the joint production of agricultural output and environmental quality will thus be characterized by the conditions

$$U_y(y,q) - C_y(y,q,w) = 0$$

$$C_q(y,q,w) = 0,$$

which imply too much agricultural output and too little environmental quality. In other words, the level of agricultural output generated by competitive markets exceeds the socially optimal level, while the level of environmental quality generated by competitive markets falls short of the socially optimal level.

In many cases, environmental effects of agriculture are associated with the use of one or more specific inputs, such as fertilizers or pesticides in the case of water pollution or farmland in the case of scenic amenities. Let z be the specific input(s) of interest and v be the associated price(s). Let e(z) represent the environmental effects of input use, where $e_z > 0$. Let the agricultural output technology be represented by a revenue function $R(p,w,z) = \max_{x,y}\{py - wx: (y,x,z) \in T\}$. Note that decreasing marginal productivity of z implies increasing marginal income foregone due to reductions in z, so that this formulation is equivalent to one characterized by increasing marginal cost of producing environmental quality. Let $S(p,e(z))$ be the social surplus accruing when the price of agricultural output is p and the level of environmental quality is $e(z)$. Under standard assumptions, $S_p = -y$, that is, the derivative of S with respect to p equals the negative of demand for agricultural output, and $S_{pp} = -\partial y/\partial p > 0$. S_e is negative for adverse environmental effects and positive for beneficial ones.

The socially optimal usage level(s) of z and agricultural output price(s) p in this case maximize net social surplus $S(p,e(z)) + R(p,w,z) - vz$ and is characterized by the necessary conditions

$$S_p(p,e(z)) + R_p(p,w,z) = 0$$

$$S_e(p,e(z))\, e_z + R_z(p,w,z) - v = 0.$$

The first condition is the market clearing condition that the quantity of agricultural output demanded ($-S_p$) equals the quantity supplied (R_p). The second condition says that the value of the marginal product of z (R_z) should equal its unit price (v) plus (minus) the marginal social cost (benefit) of environmental effects arising from use of the input(s). The marginal social cost of z equals the marginal social willingness to pay for environmental quality (S_e) times the marginal amount of environmental quality produced by z, (e_z). If markets for environmental quality are lacking, profit-maximizing farmers will equate the value of the marginal product of z with its unit price, leading to overuse of inputs that create negative environmental effects and underuse of inputs that create positive environmental effects.

2.2 Stewardship Incentives in Agriculture

Even when explicit markets for environmental quality are lacking, implicit linkages between agricultural productivity and environmental quality may give farmers incentives to provide some environmental protection. Policy discussions have traditionally referred to these incentives under the rubric of stewardship. On the production side, these incentives arise via resources that simultaneously improve both agricultural productivity and environmental quality. The canonical example is soil conservation: Preventing erosion simultaneously preserves long-run land productivity and prevents sedimentation and nutrient pollution of waterways. On the consumption side, resources may be significant sources of consumption items for farm families: Wildlife habitat allows for hunting while good water quality provides safe drinking water, fishing, and recreational opportunities.

These examples can be modeled as forms of demand for environmental quality. Using the input-oriented model, divide social surplus from consumption into three components: (1) off-farm surplus from the consumption of agricultural output, CS(p), (2) off-farm damage from use of the input of interest, D(e(z)), and (3) on-farm surplus from consumption of agricultural output and goods associated with the resource base of agriculture, L(p,e(z)), $L_p > 0$, $L_e < 0$. In the case of soil erosion, for example, z represents the erosion rate, e(z) denotes the long-run reduction in soil depth associated with erosion rate z, and D(e(z)) represents off-farm damage from sedimentation and nutrient pollution of waterways. Assuming stationary prices for agricultural output, the value of agricultural productivity in the future can be written as $L(p,e(z))$. When markets for agricultural land are well developed, $L(p,e(z))$ equals the price of farmland (Burt 1981, McConnell 1983). In the case of drinking water quality, z might represent fertilizers or pesticides and $e(z)$ the corresponding concentration in well or stream water, while $L(p,e(z))$ represents farm families' surplus from consumption. In the case of pesticides and farmer health, z would represent the toxicity, frequency of application, and/or pesticide application rate, $e(z)$ the corresponding applicator exposure, and $L(p,e(z))$ the farm household's combined demand for agricultural output and health status.

In the absence of explicit markets for environmental quality, the equilibrium is characterized by the conditions

$$CS_p + L_p + R_p = 0$$

$$L_e e_z + R_z - v = 0.$$

Farmers' unit cost of using the input z, $v\text{-}L_e e_z$, exceeds the market price v by an amount equal to the marginal reduction in land value. It is conceivable that these incentives for stewardship are strong enough to replicate the social optimum. The social optimum is characterized by the conditions

$$CS_p + L_p + R_p = 0$$

$$[L_e - D_e]e_z + R_z - v = 0.$$

The social and private optima coincide if $D_e(e(z)) = 0$ at the profit-maximizing level of z, that is, if profit-maximizing off-farm damage remains below a threshold level (Shortle and Miranowski 1987).

Alternatively, incentives for stewardship may arise from complementarity between environmental quality and agricultural output in production ($C_{yq} < 0$) or in consumption ($U_{yq} > 0$). If environmental quality and agricultural output are complements in production, farmers will have an incentive to increase environmental quality as a means of lowering the marginal cost of agricultural output. An example is altering pesticide use and other crop management practices to preserve naturally occurring beneficial insects. If demand for agricultural output is greater when environmental quality is greater (environmental quality and agricultural output are complements in consumption), then there will likely exist equilibria in which improvements in environmental quality support increased agricultural production. Organic food is perhaps the most familiar example. In either case, farmers will not treat environmental quality as costless.

2.2.1 Strength of Stewardship Incentives in Modern Agriculture

The strength of these stewardship incentives depends on the level of on-farm demand for environmental quality and on the degree of complementarity between agricultural output and environmental quality in production and consumption. These factors vary according to the type of environmental quality problem as well as across nations.

Soil Conservation. In theory, well-functioning land markets should provide farmers with sufficient incentives to conserve soil optimally in order to protect farm productivity optimally (Burt 1981; McConnell 1983), suggesting that stewardship incentives for soil conservation should be strong in developed countries. Empirical evidence from the U.S. and Australia indicates that farm land prices do reflect both past erosion and erosion potential (see for example Miranowski and Hammes 1984; Ervin and Mill 1985; Gardner and Barrows 1985; King and Sinden 1988; Palmquist and Danielson

1989) and that farmers exert greater soil conservation effort on land more vulnerable to erosion (see for example Ervin and Ervin 1982; Saliba and Bromley 1986; Norris and Batie 1987; Gould, Saupe, and Klemme 1989). In the U.S., productivity losses from erosion appear small. For example, the Natural Resource Conservation Service of the U.S. Department of Agriculture estimates that in 1992, only 19 (14) percent of U.S. cropland suffered sheet and rill (wind) erosion at rates high enough to impair productivity in any degree, while only 2 percent suffered severe erosion of either type (Natural Resource Conservation Service 1994). Simulation studies suggest erosion would reduce U.S. agricultural productivity only on the order of 3 to 4 percent over 100 years (Crosson 1986).

These measures appear insufficient to ensure adequate environmental quality, however. According to the U.S. Environmental Protection Agency, siltation and nutrients from agriculture remain the principal sources of water pollution in the U.S. despite substantial growth in the use of conservation tillage and other soil conservation measures (Economic Research Service 1997). To the best of current knowledge, off-farm damage from sedimentation and associated nutrient pollution in the U.S. is tens or hundreds of times greater than the value of erosion-induced productivity losses (Crosson 1991, Ribaudo 1986).

About 16 percent of agricultural land in developing countries is estimated to have suffered serious degradation from erosion, waterlogging, and salinization, and soil degradation is believed to have caused significant declines in agricultural productivity in a number of countries (Scherr 1998). Poorly operating credit markets and, in some countries, lack of clear property rights in land appear to be major impediments to investment in land, including soil conservation. Stewardship incentives for soil conservation thus appear to play less of a role in environmental protection in these countries than in developed countries. Deininger and Feder discuss these issues elsewhere in this Handbook.

Pesticides and Farmer/Worker Safety. Pesticide poisonings from occupational exposures are not uncommon, even in developed countries, although severe cases are relatively rare. In the U.S., for example, the incidence of reported cases of occupational exposure leading to clinically observable symptoms is about 200 per 100,000 workers. Occupational fatalities from pesticides, however, occur on average only once every few years (Levine 1991). Farmers' desire to avoid adverse health effects may be one factor in limiting occupational poisoning risk: One would expect informed farmers to take expected adverse health effects into account in making decisions about which pesticides to use, how often to apply those pesticides, pesticide application rates, and application methods. The empirical evidence available to date indicates this motive plays some role. Hedonic studies indicate a negative correlation between acute mammalian toxicity and the prices of pesticides used on corn, cotton, sorghum, and soybeans (Beach and Carlson 1993; Fernandez-Cornejo and Jans 1995). U.S. apple growers were less likely to use pesticides with higher acute mammalian toxicity, while those using more toxic pesticides applied them at lower rates (Hubbell and Carlson 1998). Corn and soybean growers in the mid-Atlantic with personal experience of adverse health effects from pesticides were more likely to use non-chemical pest control methods (Lichtenberg and Zimmerman 1999a).

Farmers' incentives for avoiding health effects from pesticide use appear less strong in developing countries. For example, Antle and Pingali (1994) found that the opportunity cost of lost work time from pesticide poisonings among rice farmers in the Philippines exceeded the increased value of rice production due to pesticide use.

Scenic Amenities, Wetlands Preservation, and Wildlife Habitat. Farmers usually live on or near their farms, making local environmental quality an item of consumption. Farmers frequently engage in outdoor recreational activities like hunting, fishing, and hiking, giving them an incentive to preserve wildlife and scenery in their local area. In these cases, farmers' demand for environmental quality can make up most or all of society's demand for environmental quality, so that farmers' use of inputs that impair environmental quality could coincide with the social optimum.

Several empirical studies have found that farmers who stated greater concern over environmental quality were more likely to use at least some farming practices that reduce runoff and erosion (Napier, Camboni, and Thraen 1986; Lynne, Shonkwiler, and Rola 1988; Amacher and Feather 1997; Weaver 1996). Lichtenberg and Zimmerman (1999b) found that mid-Atlantic farmers were willing to incur substantial extra pesticide costs on average in order to prevent pesticide leaching; concern for overall environmental quality rather than protection of human health appeared to be their principal motivation.

Other empirical findings suggest that farmers' demand for environmental quality may not be very strong. Van Kooten and Schmitz (1992) find that modest payments were insufficient to induce many Canadian farmers to preserve prairie potholes for wildfowl habitat. Weaver (1996) finds that Pennsylvania farmers who had adopted conservation practices in the past because it was the "right thing to do" currently exerted no greater conservation effort than those who had not. Beach and Carlson's (1993) hedonic price study turned up equivocal evidence: Farmers appeared willing to pay more for corn herbicides with shorter half lives and greater soil adsorption, characteristics associated with a lower likelihood of leaching into groundwater, but also for herbicides with greater water solubility, which is associated with a higher likelihood of leaching. Moreover, greater soil adsorption is associated with greater weed control as well as less leaching.

Overall, then, the literature suggests that stewardship incentives do operate in agriculture, but also that they are unlikely to satisfy society's overall demand for environmental quality from agriculture. Measures that protect farm productivity adequately typically fail to suffice for protecting broader environmental quality. Farmers' demand for environmental quality as a consumption good generally makes up only a small share of society's total demand and is thus generally inadequate to ensure attainment of socially desirable levels of environmental quality.

2.2.2 Technical Change, Farm Structure, and Stewardship

Some have argued that the emergence of agriculture as a source of environmental quality problems is linked to forms of technical change that have attenuated the importance of stewardship incentives (see for example Strange 1988). For example, the introduction of synthetic chemicals (fertilizers, pesticides) lowered the marginal value of the resource base of agriculture (soil fertility, natural populations of beneficials) and thus stewardship incentives. Moreover, they may have changed the efficient structure of farm enterprises. Strange (1988) and others have argued that, if the costs of environmental damage were fully internalized in farm decision making, smaller-scale joint crop/livestock production would be more profitable than larger-scale, specialized "industrial" farming. This idea is voiced frequently in the sustainable farming advocacy literature but has not been studied rigorously.

It has also been argued that new technologies may be the best means of reducing environmental damage from agriculture. Precision input application methods offer the greatest promise in this regard. A materials balance perspective suggests that matching input application rates more closely to crop uptake rates simultaneously tends to reduce environmental damage from such inputs. As Khanna and Zilberman (1996) point out, in most cases environmental damage is caused only by inputs that are not taken up by the crop or other organisms in the crop ecosystem.

One way to conceptualize improvements in matching application and crop ecosystem uptake rates is to use the distinction between applied and effective input use introduced by Caswell and Zilberman (1986). In their model, effective input use hz is assumed to be proportional to applied input use z. The constant of proportionality h may depend on such factors as water infiltration rates, slope, soil water-holding capacity, or soil nutrient stocks, and may be embodied in a specific delivery technology, e.g., low-pressure irrigation systems. Environmental damage is a function only of residual inputs $[1-h]z$. Hanemann, Lichtenberg, and Zilberman (1989), Caswell, Lichtenberg, and Zilberman (1990), and Dinar and Letey (1991) use this conceptualization to investigate the potential for low-volume irrigation methods to mitigate drainage problems in irrigated agriculture. They note that low-volume, pressurized delivery systems like drip can attain efficiencies as high as 95 percent, while gravity-based delivery systems typically are only about 60 percent efficient. Switching from a gravity-based to a low-volume delivery system, then, can decrease effluent production by as much as 87.5 percent. At the same time, improved matching between ecosystem demand and input application rates may result in increased crop productivity. For example, low-volume delivery systems tend to have higher yields because water delivery can be adjusted to match crop uptake rates more closely than is possible with gravity-based delivery systems. Similarly, increased crop productivity has been a major benefit of California's CIMIS evapotranspiration forecasting system (Cohen et al. 1998).

More generally, assume that crop production is a function of effective input use, which is itself an intermediate output s(z,h) produced by the input application rate z and the application efficiency h. Assume also that agricultural emissions e(z,h) are increasing in the input application rate z but decreasing in application efficiency, i.e., $e_z > 0$, $e_h < 0$. Application efficiency can be increased at a unit cost I. Increased application efficiency will result in reductions in input application rates if the two are complements, $s_{zh} > 0$. When effective input use hz is proportional to applied input use, application efficiency and application rates are complements when the elasticity of the marginal product of the effective input s is greater than one in absolute value (Caswell and Zilberman 1986). Socially optimal application efficiency and input application rates are given by

$$pf_s(h,z)s_h(h,z) - I - D_e(e(z,h)e_h(z,h)) = 0$$

$$pf_s(h,z)s_z(h,z) - v - D_e(e(z,h)e_z(z,h)) = 0,$$

while the farmer's profit-maximizing application efficiency and input application rates are given by

$$pf_s(h,z)s_h(h,z) - I = 0$$

$$pf_s(h,z)s_z(h,z) - v = 0.$$

In the absence of government intervention, farmers will tend to underinvest in application efficiency h and over-apply the input z, so that environmental damage will be greater than socially optimal.

Improvements in efficiency of this kind will tend to mitigate environmental damage on the intensive margin, that is, damage from emissions given existing cultivation patterns. But improvements in efficiency may also have extensive margin effects, e.g., changes in cropping patterns. For example, the principal impact of introducing low-volume irrigation in California was the expansion of fruit and nut cultivation onto hillsides (Caswell and Zilberman 1986, Green et al. 1996). Irrigated crop production replaced dryland pasture, so that the introduction of a water-saving technology resulted in increased aggregate water demand. Similarly, the main impact of introducing center-pivot irrigation in the northern High Plains was replacement of pasture by irrigated corn, resulting in increased risk of wind erosion (Lichtenberg 1989). Thus, it is by no means certain that improved input application efficiency will improve environmental quality.

2.3 Agriculture and the Environment: Empirical Evidence

Quantitative information about the extent of agriculture's impacts on environmental quality is remarkably scant. Even in developed countries like the U.S., most of this information is anecdotal. In particular, there are few reliable quantitative estimates of how changes in agricultural production or practices affect environmental quality.

Most empirical studies of agriculture's impacts on environmental quality use farm-level simulation models like the Universal Soil Loss Equation (USLE) for erosion and sedimentation (see for example Jacobs and Timmons 1974; Taylor and Frohberg 1977; Wade and Heady 1977; Batie and Sappington 1986; Ribaudo 1989; Ribaudo, Konyar, and Osborne 1994; Babcock et al. 1996) or the Erosion/Productivity Impact Calculator (EPIC) for groundwater (see for example Johnson, Adams, and Perry 1991; Mapp et al. 1994; Teague, Bernardo, and Mapp 1995; Helfand and House 1995). Inferences from such models are of limited value. One reason is that there is not a simple monotonic relationship between emissions at the level of an individual field and impacts on environmental quality at the ambient scale with which policy is actually concerned. Fate and transport are typically non-linear and depend on space and time in complex ways, making extrapolation of field-level emissions to ambient pollutant concentrations quite complex. Thus, while the USLE may be appropriate for describing movement of sediment from an individual field, it does not address sediment movement across fields into waterways and thus does not capture the relationship between agriculture and ambient pollution. EPIC is similarly designed to model leaching of chemicals through the crop root zone, but addresses neither lateral movement into surface water nor deep percolation into groundwater. The estimated costs of producing environmental quality appear to be quite sensitive to specification of these fate and transport relationships (Braden et al. 1989). The relationship between pollutant emissions and environmental quality impacts is similarly complex. The effects of pollutants are mediated by a variety of influences. Environmental effects frequently exhibit thresholds, that is, concentrations of pollutants at or below which there are no environmental effects due to natural degradation and/or detoxification processes. In these cases the relationship between emissions and environmental quality is not monotonic, so that models like EPIC or the

USLE are not even appropriate for measuring relative impacts of alternative policies. For example, EPIC simulations may indicate that nitrate leaching from the root zone under one policy regime is twice that under another. But under certain natural conditions (e.g., fields sufficiently far from surface water, intervening forested buffers, sufficiently small percolation), nitrate leached from the root zone may be completely removed before it reaches any body of water, and ambient pollution will be the same under both regimes.

A few studies to date have attempted to estimate agriculture/environment relationships statistically. The most noteworthy come from interdisciplinary efforts to model health effects of pesticide use on rice in the Philippines (Antle and Pingali 1994; Pingali, Marquez, and Palis 1994) and on potatoes in Ecuador (Crissman, Cole, and Carpio 1994). The Philippines project combined data on production (yield, pesticide use, use of other chemicals, family and hired labor, etc.) with data on health impairments collected by a medical team. Pingali, Marquez, and Palis (1994) link health impairments with pesticide usage patterns. Antle and Pingali (1994) develop a health impairment index that they link both to pesticide use and to the cost of rice production. The Ecuador project combined production data with information on the incidence of pesticide poisonings (Crissman, Cole, and Carpio 1994).

A few other studies have attempted to fit statistical relationships characterizing parts of the overall joint agricultural production/environmental quality technology. Anderson, Opaluch, and Sullivan (1985) obtained data on use of the insecticide aldicarb on potatoes and on concentrations of aldicarb in water in nearby wells. They were able to use these data to estimate a spatial model of aldicarb leaching into shallow well water, but were unable to obtain information on crop yields that would permit estimation of a link between well water contamination and potato productivity. Huszar (1989) combined household survey data on expenditures attributable to dust with wind erosion rates derived from the Natural Resources Inventory to estimate costs of wind erosion, but did not link wind erosion rates with agricultural production. Lichtenberg and Shapiro (1997) estimated a model linking nitrate concentrations in community water system wells with hydrological characteristics of the pertinent water-bearing formations and indicators of agricultural production (crop acreage and livestock numbers in the counties in which the wells were located) and other nitrate sources such as septic systems.

Other studies of agriculture/environment tradeoffs have used some form of detailed process modeling of agricultural production and environmental impacts, taking into account information from both crop and environmental sciences. Information from these other disciplines in specifying and parameterizing these submodels can yield important qualitative insights for policy formulation. For example, the impacts of pesticide use on farm worker safety are usually conceptualized in terms of the quantities of pesticides applied; that is, health damage from pesticides is typically modeled as a function H(x), where x indicates the total amount of pesticides applied (see for example Edwards and Langham 1976). A process model developed by Lichtenberg, Spear, and Zilberman (1994), however, indicated that the timing of pesticide application relative to harvesting operations is a critical determinant of the risk of acute poisoning from exposure to the insecticide parathion on fruit trees.

3. Heterogeneity

Because of its dependence on natural resources and natural production conditions, agriculture in most countries is heterogeneous. Crop productivity (and thus crop choice), farming practices, and input use patterns vary according to such factors as climate, topography, geology, and pest complexes. Human variability (e.g., differences in human capital across farmers) also affect crop productivity and choice. Environmental effects of agriculture vary in similar ways, partly because of variations in crop choice and cultivation methods and partly because heterogeneity in physical, chemical, and biological characteristics of the environment create differences in environmental fate and transport, exposure, and toxicity.

In this section, we explore the implications of this heterogeneity for efficient policy design. We begin by developing a conceptual model of land allocation/crop choice in land market equilibrium in a heterogeneous industry. We use this model to discuss the efficiency of the four kinds of policies used most frequently to address underprovision of environmental quality in agriculture:

1. Requiring the use of so-called best management practices;
2. Imposing restrictions on the use of specific inputs;
3. Taxing inputs associated with environmental problems; and
4. Subsidizing environmental quality measures.

We then turn to issues of implementation. We discuss the feasibility of implementing first-best policies for each of the major classes of environmental problems associated with agriculture. We then turn to policy design in cases where heterogeneity is important but regulators either cannot observe it or cannot use the information they have. We extend the basic model to encompass the design of second-best policies under hidden information (adverse selection). We conclude with a discussion of the likely applicability of such second-best policies.

The preceding list of policies has two obvious omissions: (1) taxes imposed on pollutant emissions (nitrogen runoff, pesticide leaching) rather than on inputs, and (2) pollution trading.

Emissions taxes have been used little, if at all, in agriculture. The United States has not used emissions taxes to address pollution problems generally, preferring direct regulation or, more recently, tradable permit systems. Effluent charges have been used more widely in Europe for pollution problems, but appear to have been designed to raise revenue rather than to correct externalities (see for example Cropper and Oates, 1992). Belgium and the Netherlands levy taxes on surplus nutrients from manure and fertilizers, respectively (OECD 1994), but these levies are based on theoretical rather than measured surpluses and are thus imposed on anticipated average emissions rather than measured actual emissions. Several features of agricultural pollution problems suggest that effluent taxes would be difficult to implement. First, sources of agricultural emissions tend to be numerous, widely dispersed, and difficult to identify. Second, emissions of pollutants like nitrogen or pesticides leached from fields or aerial pesticide drift are not readily observable and can be monitored only by installing expensive devices. Third, heterogeneity of production conditions and biological, physical, and chemical factors influencing the effects of emissions on ambient environmental quality makes it difficult to rely on inferences from models in order to economize on monitoring stations. As a result, monitoring emissions tends to be prohibitively costly, making emissions taxes unattractive in practice.

Pollution trading or purchasing systems have been established formally for nonpoint source nutrient problems affecting a number of watersheds throughout the U.S.

Interest in these systems is growing even though few trades have occurred to date. Heterogeneity is, of course, a necessary condition for the desirability of emissions trading—without differences in the cost of pollution reduction, agents cannot realize gains from trade. Pollution trading systems with free initial distribution of permits might be especially attractive in agriculture because such systems help mitigate the distributional effects of regulation, which tend to be pronounced in environmental policies relating to agriculture (Osteen and Kuchler 1987; Lichtenberg, Parker, and Zilberman 1988; Zilberman et al. 1991; Sunding 1996). The broader environmental economics literature has discussed a variety of issues relating to the design of such systems, including definition of permits and establishing baselines, monitoring and enforcement, market structure (the number of participants), transaction costs, initial distribution of permits, and political-economic acceptability (see Stavins 1998 for an overview, and Letson (1992) or Crutchfield, Letson, and Malik (1994) for discussion in the context of point/nonpoint trading). The infeasibility of monitoring emissions suggests that pollution trades would need to apply to inputs associated with environmental quality. Malik, Letson, and Crutchfield (1993) analyze trading between point and nonpoint sources theoretically in situations where nonpoint source emissions are random, but assume that emissions are observable. Overall, the literature lacks analyses of the features that distinguish emissions trading in agricultural situations from other methods of pricing environmental quality such as taxes or subsidies. In what follows, therefore, we treat pollution trading simply as a form of incentive.

3.1 Efficiency of Alternative Policies in a Heterogeneous Industry

A version of the input-based model modified along the lines developed by Caswell and Zilberman (1986) and by Lichtenberg (1989) is useful for discussing the relative efficiency of these alternative policy instruments. The basic model was introduced by Hochman and Zilberman (1977) as a tractable means of investigating pollution-production tradeoffs. Moffitt, Just, and Zilberman (1978), Hochman, Zilberman, and Just (1977a, 1977b), and Lichtenberg and Zilberman (1989) have applied it to problems of dairy waste management in two California river basins. Caswell and Zilberman (1986) and Lichtenberg (1989) used it to study the effects of land quality on irrigation technology choice and on cropping patterns. Caswell, Lichtenberg, and Zilberman (1990), Hanemann, Lichtenberg, and Zilberman (1989), and Shah, Zilberman, and Lichtenberg (1995) have used it to examine drainage problems in the San Joaquin Valley, California. Just and Antle (1990) discuss its use in aggregating micro-level economic and environmental models to estimate aggregate effects. Malik and Shoemaker (1993) use it to discuss targeting of cost-sharing programs for adoption of pollution-reducing agricultural practices.

Let θ represent a source or index of heterogeneity, for example, land quality (soil productivity, slope) or human capital. Let $G(\theta)$ be the cumulative distribution of θ, that is, the number of production units (acres) having at most type θ, and $g(\theta)$ be its density. For convenience, let θ be scaled such that $\theta \in [0,1]$, $G(0) = 0$, and $G(1) = N$, the total number of production units. In the short run, the number and sizes of production units can be considered fixed. Alternatively, one can assume that both technologies exhibit constant returns to scale and that the potential number of production units is limited by natural conditions such as the amount of potential farmland. In the latter case it is important to bear in mind that not all production units will necessarily be in agricultural production, so that shutdown conditions matter.

Assume also that farmers choose between two activities, which can be two different methods for producing the same crop, as in Caswell and Zilberman (1986), or different crops or crop/technology combinations as in Lichtenberg (1989). Assume that production of each is increasing and neoclassical in θ as well as use of the polluting input z_j, so that each can be represented by a revenue function $R^j(p_j,w,z_j,\theta)$. Let $\delta_1(\theta)$ denote the share of production units of type θ allocated to activity 1 and $\delta_2(\theta)$ denote the share of production units of type θ allocated to activity 2. Assume also for the moment that level of environmental quality depends only on use of z and is thus invariant with respect to production unit type θ and activity. Environmental quality can thus be written as a function of total use of z:

$$e(z) = e\left(\int_0^1 (\delta_1 z_1 + \delta_2 z_2) g(\theta) d\theta\right).$$

This restriction on e(z) will be relaxed later. Assume also that social surplus from consumption is additively separable in agricultural output (p_1,p_2) and environmental quality e(z).

The relevant decision problem is to choose p and z to maximize net social surplus plus agricultural income

$$S(p_1, p_2, e(z)) + \int_0^1 \left[\delta_1\left(R^1(p_1, w, z_1, \theta) - vz_1\right) + \delta_2\left(R^2(p_2, w, z_2, \theta) - vz_2\right)\right] g(\theta) d\theta$$

subject to the constraints that δ_1, $\delta_2 \le 1$, $\delta_1+\delta_2 \le 1$, and non-negativity constraints on δ_1 and δ_2. The necessary conditions for a maximum include

$$S_{p_j} + \int_0^1 \delta_j R_p^j(p_j, w, z_j, \theta)g(\theta)d\theta = 0, j = 1,2$$
$$S_e e_z + \left(R_z^j - v\right) = 0, j = 1,2 \forall\theta$$
$$R^j(p_j, w, z_j, \theta) - vz_j - S_e e_z z_j - \lambda_j - \lambda_0 \leq 0,$$

where λ_0 and λ_j are associated with the respective constraints $\delta_1+\delta_2 \leq 1$and $\delta_j \leq 1$. The first set of conditions is again the market-clearing conditions that aggregate demand equal aggregate supply for each activity. The second set of conditions states that the marginal net return from the use of the input z in each activity, $R_z^j - v$, should equal marginal social willingness to pay for environmental quality, $S_e e_z$. Note that optimal use of z varies across farm types and activities, since its marginal productivity varies with θ and j. The third set of conditions implies that all of each type of production unit should be allocated to the activity with the greatest social return.

The analysis that follows will be based on one of many possible equilibria. The results obtained under other possibilities are similar, but not identical. Assume that there exists a farm type $\theta_j^* > 0$ such that $R^j(p_j^*,w,z_j^*,\theta_j^*) - [v+S_e e_z]z_j^* = 0$, j = 1,2, where p_j^* and z_j^* are the optimal prices and levels of input use z. Assume without loss of generality that $\theta_1^* < \theta_2^*$, so that it is optimal for the lowest type of production unit engaged in agriculture to use activity 1. This assumption implies further that farms of types lower than θ_1^* should not engage in agricultural production, e.g., land of sufficiently low quality should not be farmed. Assume also that $R^2(p_2^*,w,z_2^*,1) - [v+S_e e_z]z_2^* > R^1(p_1^*,w,z_1^*,1) - [v+S_e e_z]z_1^* > 0$ and that $R^2_\theta > R^1_\theta$ for all θ. These latter assumptions imply the existence of a unique critical type θ_c^* defined by

$$R^1(p_1^*,w,z_j^*,\theta_c^*) - [v+S_e e_z]z_1^* = R^2(p_2^*,w,z_2^*,\theta_c^*) - [v+S_e e_z]z_2^*.$$

All production units of type $\theta_1^* \leq \theta < \theta_c^*$ should use activity 1 ($\delta_1 = 1$, $\delta_2 = 0$), while all production units of type $\theta_c^* \leq \theta \leq 1$ should use activity 2 ($\delta_1 = 0$, $\delta_2 = 1$). Thus, the remaining first-order conditions can be rewritten as

$$
\begin{aligned}
&S_{p_1} + \int_{\theta_1^*}^{\theta_c^*} R_p^1(p_1, w, z_j, \theta) g(\theta) d\theta = 0 \\
&S_{p_2} + \int_{\theta_c^*}^{1} R_p^2(p_2, w, z_j, \theta) g(\theta) d\theta = 0 \\
&S_e e_z + R_z^1 - v = 0, \theta_1^* \leq \theta < \theta_c^* \\
&S_e e_z + R_z^2 - v = 0, \theta_c^* \leq \theta \leq 1.
\end{aligned}
$$

Finally, assume for the sake of convenience that activity 1 has a smaller detrimental (or larger positive) effect on environmental quality than activity 2. If the use of z impairs environmental quality, this assumption implies activity 1 uses the input z less intensively than activity 2, that is $z_2^* > z_1^*$ for any θ. If the use of z enhances environmental quality, the opposite holds.

Without government intervention, farmers will not take environmental quality into account in choosing how intensively to use the input z or how extensively to use activities 1 and 2. If the use of z impairs environmental quality, then farmers of every production type will use too much z in both activities ($z_j^o > z_j^*$ for all θ). Returns to agricultural production will be higher than socially optimal (see Figure 1). Both the breakeven production type below which agricultural production will not occur and the critical production type at which activity 2 becomes more profitable than activity 1 will be lower than socially optimal ($\theta_1^o < \theta_1^*, \theta_c^o < \theta_c^*$). As a result, production units of types $\theta_c^o < \theta < \theta_c^*$ and $\theta_1^o \leq \theta < \theta_1^*$ will be allocated suboptimally. Output of activity 2 will be higher than socially optimal both because of the increased intensity of use of z and because of the increased extent to which activity 1 is used. The price of the output of activity 2 will consequently be lower than socially optimal. Output of activity 1 may be higher or lower than socially optimal because increases in output due to more intensive use of z and more extensive use of activity 1 on low type production units may be counterbalanced by decreases in output due to the decreased extent to which activity 1 is used on higher type production units.

If the use of z enhances environmental quality, as in the case of scenic amenities from farmland, the distortion caused by ignoring environmental quality will involve less intensive use of z and less extensive use of activity 1 ($\theta_1^o > \theta_1^*$, $\theta_c^o < \theta_c^*$) than is socially optimal.

3.1.1 Best Practice Requirements

One approach to remedying the underprovision of environmental quality in agriculture is to mandate the use of practices that are less profitable but provide more environmental quality. For example, storage of animal wastes prevents runoff during storms and permits expanded use of manure as a substitute for chemical fertilizers during the growing season. Requiring installation of manure storage facilities and restrictions on spreading manure have been used to address runoff of animal wastes in Denmark (Dubgaard 1990) and some localities in the U.S. (Moffitt, Just, and Zilberman 1978). In Australia, local districts require landowners to carry out specific erosion control measures, while Australia and Japan restrict land clearing (OECD 1994). In the U.S. and Europe, leaching of pesticides into groundwater and aerial drift of pesticides have been addressed primarily through restrictions on mixing, loading, and application methods (Lichtenberg 1992; OECD 1994). Zoning land for agricultural use, used in the U.S. and other countries, effectively requires landowners to maintain certain forms of open space.

For convenience, consider the case in which the use of z has negative effects on environmental quality while activities 1 and 2 both produce the same kind of output but with different input intensities. Social surplus from the consumption of agricultural output and environmental quality is thus S(p,e(z)). Imposition of a best practice requirement corresponds to mandating the use of the less-polluting activity (activity 1) while prohibiting the use of the more polluting activity (activity 2).

Since $R^2\text{-}vz_2^o > R^1\text{-}vz_1^o$ and $z_2 > z_1$ for production types $\theta \geq \theta_c^o$, agricultural output from activity 2 is greater than output from activity 1 for production types $\theta \geq \theta_c^o$. Thus, the best practice requirement will result in a reduction of agricultural output and an increase in its price, although neither will necessarily attain its socially optimal level. The minimum production type engaged in agricultural production will remain θ_1^o. Use of the input z will fall in production types $\theta \geq \theta_c^o$ due to the shift from activity 2 to activity 1 and will remain the same in production types $\theta_1^o \leq \theta < \theta_c^o$. Environmental quality will improve, although it will not necessarily attain its socially optimal level. Overall, best practice requirements cannot replicate a first-best allocation of resources because they operate only on the extensive margin (altering choice of activity among production types), and then only in a relatively crude way.

3.1.2 Input Use Restrictions

A second policy approach is placing restrictions on the use of inputs associated with environmental quality degradation or enhancement. Most developed countries prohibit the sale of pesticides for use in situations (i.e., specific crops or locales) where they are thought to have excessive negative environmental or human health effects (OECD 1994). The use of sewage sludge as fertilizer may be limited or prohibited in areas where treated fields are thought to breed mosquitoes or be a source of bacterial contamination of water bodies. Fertilizer application may be limited in areas with severe problems of nitrate contamination.

Consider again the case in which the use of the input z has negative effects on environmental quality and uniform restrictions on its use are imposed to attain the socially optimal level of environmental quality. There will exist critical types $\bar{\theta}_1$ and $\bar{\theta}_2$ at which these constraints become binding for activities 1 and 2, respectively. If $R_{z\theta} > 0$, so that optimal use of z is greater on higher-type production units, then the use of z will be unaffected on types below these critical types ($z_1^r = z_1^o$, $\theta_1^r \leq \theta < \bar{\theta}_1$ and $z_2^r = z_2^o$, $\theta_c^r \leq \theta < \bar{\theta}_2$) and will equal the constrained level on all other types ($z_1^r = \bar{z}_1$, $\bar{\theta}_1 \leq \theta < \theta_c^r$ and $z_2^r = \bar{z}_2$, $\bar{\theta}_2 \leq \theta \leq 1$). The lowest type of production unit engaged in agriculture will remain unchanged ($\theta_1^r = \theta_1^o$). The lowest type of production unit using activity 2 will fall ($\theta_c^r < \theta_c^o$) because the constraint on use of z will be binding for activity 1 but not activity 2 on production unit type θ_c^o, making activity 1 less profitable than activity 2 for units of that type. Agricultural output from both activities will decline and their prices will rise, but neither is likely to attain its socially optimal level.

As is well known, uniform restrictions fail to replicate social optima because they fail to take heterogeneity into account. However, the size of efficiency loss associated with the use of uniform restrictions depends on the degree of heterogeneity. For example, Moffitt, Just, and Zilberman (1978) find that the minimum costs of managing dairy waste to meet water quality standards in the Santa Ana River Basin in California were over 20 percent lower than the costs associated with standards requiring the same disposal area per cow for all dairies. Fleming and Adams (1997) find that farmers' returns were about 13 percent higher under a spatially differentiated tax than under a uniform tax when both taxes achieved the same level of groundwater quality. In two of the four soil types considered, there was no difference in farm returns. Lichtenberg, Zilberman, and Bogen (1989) find that the cost of meeting uniform standards for the carcinogenic pesticide DBCP in drinking water in all wells of a multiple-well system in California exceeded the minimum cost by only 4 to 6 percent. Helfand and House (1995) find that the welfare cost of meeting nitrogen standards in groundwater in California was only about 2 percent higher under a uniform tax than under taxes differentiated by soil type.

3.1.3 Linear Input Taxes and Subsidies

Taxes on polluting agricultural inputs have received only limited use for addressing environmental quality problems in agriculture. In the United States, the state of Iowa imposed a relatively small tax on fertilizer whose primary purpose was to fund development and dissemination of best farming practices rather than influence fertilizer consumption. Austria, Germany, Finland, Denmark, Sweden, Spain, and the United Kingdom tax fertilizers and pesticides, although it is not clear whether they set tax rates according to environmental criteria (Oskam, Viftigschild, and Graveland 1997).

Subsidies for inputs thought to enhance environmental quality are more widely used. Sweden, Austria, Switzerland, and the United Kingdom pay farmers to maintain amenities such as wildlife habitat and mountain landscapes. Most developed countries use subsidies to keep land in agriculture. Many national and local governments also provide implicit subsidies by taxing agricultural land at lower rates than land in other uses (OECD 1994).

It is apparent from the necessary conditions for a social optimum that imposition of a constant per-unit tax (subsidy) equal to $S_e e_z$ levied on a polluting (environmental quality enhancing) input z will induce farmers of all types to choose optimally the level of z and the kind of agricultural activity. Thus, when environmental quality is sensitive only to input use z and not to farm type θ, a uniform tax (subsidy) on the polluting (environmental quality enhancing) input will achieve a social optimum even when heterogeneous conditions lead farmers to choose different crops, different farming practices, and different input mixes. The result follows because the tax (subsidy) gives farmers the appropriate incentives on both the intensive and extensive margins.

An alternative to taxing polluting input use (or emissions) is to tax farmers on the basis of observed ambient environmental quality. When environmental quality is sensitive only to input use z and not to farm type θ, a tax S_e on observed ambient environmental quality e(z) will induce farmers of all types to choose optimally the level of z and the kind of agricultural activity (Griffin and Bromley 1982).

In contrast, a constant per-unit subsidy on reductions in the use of a polluting input will not achieve a social optimum and may even worsen environmental quality. In the best case, one could subsidize reductions in the use of z from (known) profit-maximizing levels at a rate equal to the marginal value of environmental quality enhancement in the social optimum, in other words, offer a subsidy of the form $S_e e_z [z_j(\theta) - z_j^o(\theta)]$. Such a subsidy will lead to socially optimal levels of z on most types of production units, but not all. The subsidy will lower the minimum type of production unit engaged in farming,

$$\frac{\partial q_1^o}{\partial t} = -\frac{z_1^o - z_1^*}{R_q^1} < 0,$$

where t is the subsidy rate. The use of z will thus increase on all production units of types $\theta_1^s \leq \theta \leq \theta_1^o$, where θ_1^s is the minimum type of production unit engaged in agriculture. The subsidy will also change the critical type θ_c^o by

$$\frac{\partial q_c^o}{\partial t} = -\frac{(z_2^o - z_2^*) - (z_1^o - z_1^*)}{R_q^2 - R_q^1}.$$

If the distortion in the use of z is greater for activity 2 than activity 1 $(z_2^o - z_2^* > z_1^o - z_1^* > 0)$, the subsidy will lower the critical quality, and increase the use of z on all production units of types $\theta_c^s \leq \theta \leq \theta_c^o$, where θ_c^s is the critical minimum type engaged in activity 2. If the distortion is greater for activity 1 than activity 2, the reverse holds. Since the subsidy increases the use of z for some types of production unit, it may actually lead to increased total use of z and thus decreased environmental quality. This result, of course, corresponds to the well-known result that subsidies for pollution reduction may increase pollution in the long run (see for example Cropper and Oates 1992). The results and derivation also correspond closely to Caswell and Zilberman's (1986) demonstration that the introduction of a water-saving irrigation technology may not result in aggregate water savings. If activities 1 and 2 produce different crops, then these shifts on the extensive margin will lead to a suboptimal mix of agricultural products as well as excessively low environmental quality.

3.1.4 Best Practice Subsidies

Subsidies for best practices are used widely to addressing environmental quality problems in agriculture, at least in the United States and Europe. The principal approach taken toward water quality issues in the United States has been to provide subsidies for best management practices thought to reduce erosion and runoff and for maintaining highly erodible land in conservation uses (for surveys of programs see Reichelderfer 1990 or OECD 1994). The Conservation Reserve Program (CRP), which features paid retirement of land thought to contribute excessively to environmental problems, is arguably the most significant environmental initiative in agriculture in the United States. Conceptually, the CRP is a narrowly targeted form of best practice subsidy in which grassland or forestry is the socially optimal use of land. The European Union, New Zealand, Japan, and Turkey have similar programs. Most of these countries also provide subsidies in the form of cost sharing for farmers adopting approved practices, usually for readily observable actions such as installation of structures, establishment of perennial crops, or planting winter cover crops. Free extension service advice regarding such practices, a common component of these programs, can also be considered a form of subsidy.

In the case where use of the input z impairs environmental quality, such best practice subsidies can be modeled as a lump-sum subsidy to farmers choosing to use activity 1. Such a subsidy will not change farmers' incentives to use z on the intensive margin, but it will affect farmers' choices regarding whether to engage in farming and which activity to select. Its effects are shown in Figure 2. Like a subsidy on reductions in the use of z, a lump-sum payment T to users of practice 1 will lower the minimum type of production unit engaged in farming ($\theta_1^T < \theta_1^o$) and raise the critical type choosing activity 2 ($\theta_c^T > \theta_c^o$). The impact on the use of z is thus ambiguous: Farmers switching from activity 2 to activity 1 will use less of z, while those switching to farming from non-agricultural activities will use more.

Land retirement programs like the CRP feature paid diversion of land thought to contribute excessively to environmental problems. An optimal program of this type will pay farmers to retire land that is privately profitable but socially inefficient to farm. Formally, such policies should address types of production units characterized by the conditions

$$R^j - vz_j^o > 0$$
$$R^j - (v - S_e e_z)z_j^* < 0,$$

which apply to production units of types $\theta_1^o \le \theta < \theta_1^*$, i.e., an efficiently structured program like the CRP should pay farmers R^1-vz_1 to retire land of types $\theta_1^o \le \theta < \theta_1^*$. Agricultural production and input use will be socially optimal in units of these types, but will be unaffected in all other types. Thus, a land retirement program like the CRP is capable of generating a full social optimum only if environmental quality is unaffected by input use in production units of other types.

In sum, best practice subsidies are generally incapable of achieving social optima because they affect farmers' decisions only on the extensive margin. Best practice subsidies that reduce the use of polluting inputs sufficiently to achieve a socially optimal level of environmental quality will simultaneously produce socially suboptimal levels of agricultural output and output prices. Moreover, subsidies of this kind tend to be excessively costly due to targeting problems, since the subsidy affects only the decisions of a few types of production units even when offered to all users of a practice. Land retirement programs like the CRP can be structured to generate socially optimal production in production units of types that should not be engaged in agricultural production, but cannot affect input use in units of types remaining in production. They can be efficient within these limits but cannot replicate social optima by themselves. In other words, land retirement programs like the CRP can be efficient only when coupled with other policies that influence input use on the intensive margin.

3.2 Heterogeneity and Implementability

It has been assumed so far that the production of environmental quality depends only on the amount of the input z used. But in many cases environmental quality effects depend on both the type of production unit and the agricultural activity. For example, leaching of fertilizers tends to be greater on sandier soils and on irrigated crops. Similarly, some crops have greater fertilizer requirements or take up nutrients less efficiently than others. Erosion and thus sediment damage and nutrient pollution tend to be greater on land with steeper slopes and for row crops. Fish kills from pesticide drift are more common when fields are closer to streams and lakes. Cropland irrigated using drip systems generates less runoff and drainage than cropland irrigated with gravity systems. Farmland located close to urban areas provides more scenic amenities.

Suppose that environmental quality does vary by type of production unit and production activity, so that environmental quality provided by a production unit of type θ in activity j is $e^j(z_j,\theta)$ and total environmental quality is

$$e(z) = \int_0^1 \left[\delta_1 e^1(z_1(\theta),\theta) + \delta_2 e^2(z_2(\theta),\theta)\right] g(\theta) d\theta .$$

The necessary conditions for a social optimum in this case are

$$\begin{aligned} & S_{p_j} + \int_0^1 \delta_j R_p^j(p_j, w, z_j, \theta) g(\theta) d\theta = 0 \\ & R_z^j - v + S_e e_z^j = 0 \quad \forall \theta \\ & R^j(p_j, w, z_j, \theta) - v - S_e e^j - \lambda_j - \lambda_0 \leq 0. \end{aligned}$$

The optimal input tax in the present case is $S_e e_z^1$ if activity 1 is more profitable and $S_e e_z^2$ if activity 2 is more profitable and thus varies according to both production activity and type of production unit (see also Griffin and Bromley 1982; Segerson 1988). In contrast to the results of section 4.2.3, therefore, the social optimum cannot be implemented by a uniform tax on inputs that impair environmental quality or by a uniform subsidy on inputs that enhance environmental quality. Suppose for example that the polluting input is nitrogen fertilizer, that θ indexes land quality, that the two activities using fertilizer are corn and wheat production, and that nitrogen leaching into surface water causes damage from eutrophication. Then a nitrogen tax varying according to land quality and crop produced will induce farmers to apply nitrogen and to allocate their land optimally between corn, wheat, and pasture. If nitrogen leaching is lower on higher quality land (e.g., higher quality land has higher water holding and/or cation exchange capacity), then the optimal nitrogen tax on both corn and wheat should be lower on fields of higher quality. If corn uses nitrogen more efficiently than wheat (e.g., corn takes up a higher percentage of applied nitrogen than wheat), then the nitrogen tax on corn should be less than the nitrogen tax on wheat. Put another way, a tax on nitrogen that is not differentiated according to land quality and crop will be suboptimal on both the intensive and extensive margins: It will fail to induce farmers to apply nitrogen optimally on each crop (the intensive margin) and will also fail to induce farmers to allocate land optimally between corn and wheat (the extensive margin).

The social optimum can be implemented by a uniform tax equal to S_e levied on emissions e^j, as Griffin and Bromley (1982) note. But monitoring emissions tends to be excessively costly or infeasible. Regulators may therefore be unable (or find it too costly) to observe production unit type, for example, land quality or human capital, either directly or by inference from observed emissions, production practices or yield. For example, the same crop may be grown in a number of different types of land or by farmers with different levels of expertise. Alternatively, the government may observe farmers' types and actions, but may lack the legal authority to impose policies that discriminate among farmers on the basis of either (Chambers 1992). The U.S. Department of Agriculture has offices in almost every county in the United States, for example. It has mapped soils extensively. Commodity programs in the past required farmers to report cropping patterns. USDA surveys estimate crop yields and input usage. Data on weather is available from numerous stations located in reasonable proximity to most farms. One would thus expect USDA to have the capacity to monitor the use of inputs affecting environmental quality either directly or by inference. Yet it may still be prevented from implementing differential taxes needed to achieve a social optimum. For example, imposing on fertilizer used on corn and wheat or on water applied with different irrigation methods may be prohibited by law, either specifically by statute or by judicial interpretation of common law. Heterogeneity can thus be a source of regulatory failure, that is, of the inability of regulation to achieve a socially optimal allocation of resources.

3.2.1 Implementability of First-Best Taxes and Subsidies

The extent to which it is feasible to implement a social optimum (or a reasonable approximation thereto) varies according to the type of environmental quality problem.

Fertilizers. As noted above, environmental damage from fertilizers typically varies according to both cropping pattern and attributes of land quality such as slope and soil texture. Nutrient pollution of surface and ground waters tends to be greater on greater slopes, on land closer to streams, and on sandier soils. Crops differ in efficiency of nutrient uptake and thus in the supply of residual nutrients available for runoff and leaching. Crop rotations featuring winter cereal crops often generate less nutrient pollution because the cereal crops absorb significant shares of residual nutrients.

Implementing differential taxes on fertilizers would be difficult. Farmers use the same fertilizer formulations on different crops and different types of land, making it infeasible to impose differential taxes at the point of sale. Variations in yield are caused by numerous factors, including ones that are not easily observed like seed variety, pest infestation levels, and microclimate, making it difficult to infer fertilizer application rates from observed yields. One would thus expect it to be difficult to implement first-best taxes on fertilizers.

The situation may differ in areas suffering from drainage problems. Emissions of leached fertilizers (more generally, of nutrients or heavy metals occurring naturally in soils) in such areas come from water pumped from subsurface drains. In essence, nutrient emissions in such areas come from point sources that can be monitored. The energy required for pumping is proportional to the volume of water pumped. When pumps are powered by electricity, the volume of effluent can be inferred from electricity consumption, so that taxes on effluent can be levied on electricity use. Soil maps can be used to adjust such taxes for differences in emissions due to soil quality or the presence of naturally occurring nutrients or heavy metals. It would be more difficult to make adjustments for cropping patterns, however. Further difficulties are likely to arise in areas in which subsurface drains capture leachate from neighboring farms as well as from the farm on which the drains are located. Such situations are common in areas with perched water tables, such as the west side of the San Joaquin Valley, California (Loehman and Dinar 1994). Thus, implementation of first-best taxes is likely to be feasible in areas where drainage flows across farm boundaries are small and where there is little variation in cropping patterns or pollutant emissions across crops.

Livestock Wastes. Environmental damage from livestock wastes typically varies according to the location of the livestock operation, the extent to which animals are concentrated, and the type of livestock involved. Nutrient pollution from livestock wastes tends to be greater on land near streams, near feedlots, and on sandier soils. Poultry litter and hog waste have greater concentrations of nutrients than cow manure. Livestock wastes present a mixed case for implementability. In some cases, livestock operations can be treated as point sources and thus regulated differentially. Moffitt, Just, and Zilberman (1978), Hochman, Zilberman, and Just (1977a, 1977b), and Lichtenberg and Zilberman (1989) investigate cases of surface water pollution from dairy wastes left on pastures or spread on disposal areas. Dairies can be treated as point sources in such cases. In other situations, livestock wastes are spread on cropland as fertilizer. In such situations, differential taxation of livestock wastes would be characterized by the implementation problems discussed above for fertilizers.

Erosion. Environmental damage from erosion (sedimentation, nutrient pollution) typically varies according to topography, soil characteristics, location (e.g., proximity to streams), crop choice, and farm production practices, all of which should be considered observable by USDA and extension personnel or from soil survey data. (In contrast to variable input use, most farm practices with significant effects on erosion are easily observed.) In principle, then, it should be feasible to levy on farmland first-best taxes that are adjusted for crop type and farming practices. Legal restrictions likely constitute the most significant barrier to doing so.

Pesticides. Environmental damage from pesticides typically varies according to the formulation (and thus application method) used and according to the location of the field to which pesticides are applied. It should be feasible to impose differential taxes on different formulations of any given pesticide active ingredient (e.g., liquid versus granular), although additional measures may be required in some cases to prevent dealers or farmers from evading higher taxes by reformulating pesticides themselves. Similarly, it should be feasible to impose differential taxes on pesticides purchased in different regions, although it may be necessary to simultaneously implement enforcement measures to limit smuggling. Existing pesticide regulation provides precedents for differential treatment of this kind. The U.S. Environmental Protection Agency (EPA) banned all uses of granular formulations of the insecticide carbofuran due to concerns over bird kills but did not take regulatory action against other formulations. EPA has similarly cancelled the registrations of numerous chemicals in specific states or growing regions while permitting legal use to continue in other areas.

Scenic Amenities, Wetlands Preservation, and Wildlife Habitat. Environmental quality enhancement from farmland preservation or other land use restrictions (wetlands preservation, wildlife habitat, paid conservation set-asides) depends primarily on the location of the farm and secondarily on agricultural production practices. One would expect demand for scenic amenities to be greater for farms closer to urban areas with larger populations. Concentrated livestock operations tend to have negative impacts on environmental quality due to odor. How much the public is willing to pay for the different kinds of scenery associated with different crops has not been investigated. Estimates of public willingness to pay for open space and scenery as functions of farm location and farm characteristics can be used to derive values for the purchase of development rights or agricultural easements or rental of land for wetlands preservation, wildlife habitat, or erosion control. Implementing a first-best thus seems feasible for all of these environmental services. However, lack of information about differences in returns to farming due to human capital or about farmers' subjective assessments of future returns to farming and of the price of their land for development may hinder implementation of first-best policies, especially in programs that are local in scope and thus feature limited numbers of potential participants (e.g., preserving farmland for scenic amenities). Programs aimed at environmental quality problems on a larger scale can exploit competition among potential participants to achieve a first-best. From this perspective, for example, the decision to administer the CRP on a county basis is inconsistent with a goal of minimizing the cost of achieving environmental quality improvements.

3.2.2 Second-Best Implementation Under Hidden Information

When the government is unable to observe (or use its knowledge about) heterogeneity in the conditions governing the production of agricultural output and environmental quality, it is forced to consider second-best policies. One example of such policies is non-linear taxes on polluting inputs, in which the tax rate varies according to the amount of the input purchased. Other examples include auctions for CRP or wetlands reserve rentals, farmland easement purchases, and sale of development rights, in which the price paid for a unit of land varies according to the size of the parcel offered.

The derivation of such second-best policies is performed under the assumption that the government has perfect information about the structure of the farm economy and of environmental quality but cannot distinguish between individual farmers. In other words, the government knows agricultural production technology and thus $R^j(p_j,w,z_j,\theta)$, emissions and thus $e^j(z_j,\theta)$, and the distribution of types of production units $G(\theta)$, but does not know the type θ of any individual farm operation.

Second-best taxes and subsidies of this kind can be implemented using a revelation mechanism that induces farmers to report their type θ truthfully to the government in return for a payment (positive or negative) that depends on the type reported (for a detailed exposition see Chambers elsewhere in this Handbook). Suppose for example that the government wants to maximize social surplus from the consumption of agricultural output and environmental quality $S(p_1,p_2,e(z))$ where the input z has a negative effect on environmental quality, $S_e < 0$, plus revenue from a non-linear tax $t(\tilde{q})$ where the tax payment depends on the type reported to the government $\tilde{q}$. (If revenue from the tax reduces the need to levy other distortionary taxes, then the social value of each dollar of revenue from the tax on the polluting input may be worth more than a dollar.)

This problem is the same in most respects as the case investigated by Spulber (1988) in which a regulator attempts to maximize surplus from consumption of a product less the social value of pollution damage when the costs of pollution control vary across firms and when the regulator knows the distribution of firm types but cannot observe or infer the type of any individual firm. It differs in two important respects. First, it considers multiple outputs (the two agricultural activities) and thus examines switches among outputs on the extensive margin. Second, it considers the possibility of shutdown, that is, of zero production by some types.

The amount earned by a farmer operating a production unit of type θ who reports type $\tilde{q}$ is

$$d_1[R^1(p_1,w,z_1(\tilde{q}),q) - vz_1(\tilde{q})] + d_2[R^2(p_2,w,z_2(\tilde{q}),q) - vz_2(\tilde{q})] - t(\tilde{q}) .$$

If truthful revelation is optimal for the farmer, earnings for a farmer of type θ must be at a maximum when the farmer reports type $\tilde{q}$. A necessary condition is thus that

$$\left(\delta_1[R_z^1 - v] + \delta_2[R_z^2 - v]\right)z_\theta - t_\theta = 0 ,$$

which implies that z_θ and t_θ have the same sign. The incentive compatibility condition implies that if optimal use of the polluting input z is higher (lower) on production units of higher (lower) types, then the total tax payment will be higher (lower) as well. Note that this does not imply that the tax rate per unit of input rises as use of the input rises.

This condition will be sufficient for a maximum when

$$\frac{\partial}{\partial\tilde{\theta}}\left\{\left(\delta_1[R_z^1 - v] + \delta_2[R_z^2 - v]\right)z_\theta - t_\theta\right\} + \left(\delta_1 R_{z\theta}^1 + \delta_2 R_{z\theta}^2\right)z_\theta \leq 0$$

(see for example Guesnerie and Laffont 1984 or Fudenberg and Tirole 1991). Since the first term is non-positive by assumption, sufficiency is assured if z_θ and $R^j_{z\theta}$ have the same sign. If the marginal product of z is higher on units with higher production types, as has been assumed in the previous section, then a second-best tax should ensure that use of z is higher on higher-type production units. If this monotonicity condition is not met for some types of production unit, then it will be optimal to impose taxes that ensure that all units of those types use the same amount of z; for these types, the second-best tax will achieve a pooling equilibrium.

Under the assumptions of the preceding section, the social optimum has the following characteristics. Activity 1 uses less of the polluting input z than activity 2. It is socially optimal to use activity 1 on lower-type production units. There exist critical types θ_1* and θ_c* such that agricultural production should not occur for types $\theta < \theta_1$*. Activity 1 is socially optimal for types $\theta_1* \le \theta < \theta_c$*, and activity 2 is socially optimal for types $\theta \ge \theta_c$*.

Knowing the agricultural production technology means that the government knows θ_1*. It can thus choose a tax schedule such that farmers with production units of type θ_1* earn zero quasirent, that is

$$R^1(p_1,w,z_1,\theta_1*) - vz_1 - t(\theta_1*) = 0$$

when z_1 is chosen to maximize the farmer's earnings. Integrating the incentive compatibility condition implies that

$$t(\theta) = \left(\delta_1[R^1(\theta) - vz_1(\theta)] + \delta_2[R^2(\theta) - vz_2(\theta)]\right) - \int_{\theta_1^*}^{\theta}[\delta_1 R_\theta^1 + \delta_2 R_\theta^2]ds.$$

Note that for types $\theta < \theta_1$*, the optimal tax is

$$t(\theta) = R^1(\theta) - vz_1(\theta) + \int_{\theta}^{\theta_1^*} R_\theta^1 ds > R^1(\theta) - vz_1(\theta),$$

which implies that the optimal tax will force out of production all units of types less than the socially optimal minimum θ_1*. Thus, a second-best tax is capable of implementing a social optimum on at least the lower-end extensive margin.

The government implements the second-best program by selling a farmer revealing herself to be type $\tilde{q}$ the right to purchase an amount of the polluting input z at the market unit price v in return for a payment $t(\tilde{q})$. Its objective is to choose z and t to maximize social surplus from consumption of agricultural output and environmental quality plus tax payments:

$$S\left(p_1, p_2, \int_{\theta_1^*}^{1}[\delta_1 e^1(z_1) + \delta_2 e^2(z_z)]g(\theta)d\theta\right) + \int_{\theta_1^*}^{1} t(\theta)g(\theta)d\theta.$$

(The extension to the case where tax payments displace distortionary taxes is straightforward.) Assuming truth-telling implies that the objective function can be expressed (after integration by parts) as

$$S\left(p_1, p_2, \int_{\theta_1^*}^{1}[\delta_1 e^1(z_1) + \delta_2 e^2(z_z)]g(\theta)d\theta\right) +$$
$$\int_{\theta_1^*}^{1}\left\{\delta_1[R^1(p_1, w, z_1, \theta)] + \delta_2[R^2(p_2, w, z_2, \theta)] - \frac{N - G(\theta)}{g(\theta)}[\delta_1 R_\theta^1 + \delta_2 R_\theta^2]\right\}g(\theta)d\theta.$$

The necessary conditions for a maximum are

$$S_{p_{1j}} + \int_{\theta_1^*}^{\theta_c^m}\left\{R_p^1 - \frac{N - G(\theta)}{g(\theta)}R_{\theta p}^1\right\}g(\theta)d\theta = 0$$
$$S_{p_2} + \int_{\theta_c^m}^{1}\left\{R_p^2 - \frac{N - G(\theta)}{g(\theta)}R_{\theta p}^2\right\}g(\theta)d\theta = 0$$
$$R_z^j - v + S_e e_z^j - \frac{N - G(\theta)}{g(\theta)}R_{\theta z}^j = 0$$

where θ_c^m is defined by the equation

$$\left[R^1(q_c^m) - vz_1(q_c^m) + S_e e^1(q_c^m)\right] - \left[R^2(q_c^m) - vz_2(q_c^m) + S_e e^2(q_c^m)\right] -$$
$$\frac{N - G(q)}{g(q)}\left[R_q^1 - R_q^2\right] = 0.$$

These conditions are sufficient if the monotonicity condition holds. The monotonicity condition can be checked by differentiating the system first-order conditions with respect to θ and solving for z_θ. As usual, monotonicity of the hazard rate [N-G(θ)]/g(θ) is necessary but not sufficient for the second-order condition for truth-telling to hold.

First, note that at θ_c* the first two terms of the equation defining θ_c^m are equal while the third term is negative, indicating that $\theta_c^* > \theta_c^m$. Thus, the optimal second-best tax will induce production units of types $\theta_c^m \leq \theta < \theta_c^*$ to use activity 2 instead of the socially optimal activity 1.

Second, if $R_{z\theta} > 0$, that is, the marginal product of z is greater on higher-type units, then units of types $\theta_1^* \leq \theta < \theta_c^m$ and $\theta_c^* \leq \theta < 1$ will use less of the input z than is socially optimal. Units of type 1 will use the socially optimal level of z, that is, there is no distortion of input use or production "at the top." (Recall that G(1) = N so that the optimal second-best tax equals the optimal first-best tax for units of type 1.) The use of z on units $\theta_c^m \leq \theta < \theta_c^*$ will be greater than socially optimal because activity 2 uses z more intensively than activity 1, however, so that the optimal second-best tax may result in either more or less use of z than is socially optimal.

Third, the second-best tax will result in less agricultural output from both activities and more environmental quality than is socially optimal on the intensive margin. (Spulber (1988) derives a similar result.) The output of activity 1 will always be less than socially optimal. Both the output of activity 2 and environmental quality may be more or less than socially optimal because the increased use of z and increased output by production units of types $\theta_c^m \leq \theta < \theta_c^*$ will counteract the reduced use of z and decreased output of activity 2 on other types.

3.2.3 Implementability of Second-Best Mechanisms Under Hidden Information

Most discussions of implementability of second-best mechanisms under hidden information focus on the monotonicity condition discussed in the preceding section. But several other conditions are necessary for these second-best mechanisms to be enforceable. Two in particular—compliance monitoring and secondary markets—are crucial but frequently overlooked.

First, the government must be able to observe compliance in some way. If it imposes a tax on the use of a polluting input, it must be able to monitor tax collections to ensure that quantity discounts or premia are actually applied to farmers' purchases. If the second-best tax involves quantity premia, the government must be able to monitor cumulative purchases to ensure that farmers do not simply avoid higher tax rates through multiple purchases of smaller quantities. If the government offers a subsidy for conservation practices or for open space, it must be able to observe whether farmers actually carry out their side of the bargain. If the government offers a payment in return for conservation effort, it must be able to observe that effort (the equivalent of a time sheet for labor) or infer it from observable outcomes.

Second, second-best mechanisms are enforceable only when secondary markets are infeasible, e.g., when the input z cannot be stored or resold or when users of z cannot collude (for example, by forming purchasing cooperatives). Otherwise, if the optimal second-best tax rate featured discounts for larger purchases, farmers using smaller amounts of the input z would find it optimal to purchase larger amounts than would be profitable for a single season and either store the excess for the following season or sell the excess to other users. The formation of purchasing cooperatives would accomplish the same end. Similarly, if the optimal second-best tax rate featured price premia for larger purchases, farmers using larger amounts of z would find it profitable to make multiple purchases of smaller lots.

These considerations suggest that second-best mechanisms will not generally be viable methods of handling environmental problems associated with the use of variable inputs like chemicals (fertilizers, pesticides). Second-best taxes on such would involve extensive reporting to ensure that the appropriate tax be assessed on farmers' cumulative purchases from all dealers. If the second-best tax involved quantity premia, periodic inspections would be needed to limit the use of straw buyers. Even so, secondary markets for chemicals can emerge readily. Chemicals are storable and easily repackaged. Moreover, some farmers already engage in contract chemical application for others, making enforcement more complex. The resulting enforcement problems could well eclipse those traditionally associated with moonshine.

Subsidies for limiting chemical use like the "green payments" program suggested by Wu and Babcock (1995, 1996) are similarly unenforceable without extensive, intrusive government inspection. Monitoring cumulative purchases presents the difficulties noted above. Moreover, because chemicals are storable, their (single-season) use cannot be inferred reliably from purchases. Actual applications can be adjusted on a continuing basis, so compliance monitoring would require virtual continuous observation of farm operations. In the absence of effective monitoring, there is nothing to prevent farmers from accepting payments and then simply applying chemicals at profit-maximizing levels. Furthermore, subsidies for reductions in polluting activities may have undesirable extensive margin effects like crop switching or bringing extra land into cultivation.

Second-best mechanisms are more likely to be enforceable in cases involving the use of durable inputs such as conservation structures (terraces), perennial crops (stripcropping, filter strips, riparian buffers, reforestation), or the maintenance of environmental amenities (open space, scenic amenities, wildlife habitat). In these cases, inspection at relatively infrequent intervals suffices to assess compliance. Smith (1995), for example, considers the design of a bidding system for enrolling land in the CRP in cases where farmers possess private information about the profitability of the land they offer. His model is more applicable to programs that are local in scope, such as those aimed at preserving farmland as open space, since administration of the CRP on a national basis would create sufficient competition among potential participants to allow the government to avoid paying information rents.

Mechanisms involving the use of durables are not necessarily self-enforcing, however. Livestock wastes are a case in which further enforcement is likely necessary. The government can observe at relatively low cost whether farmers build and maintain waste storage structures, but monitoring waste disposal is more costly, because manure can be spread at almost any time. It is possible that farmers will build waste storage structures yet continue to spread manure for disposal rather than use it as a fertilizer substitute (see Dubgaard (1990) for an example from Denmark).

The preceding discussion suggests that the greatest scope for the application of hidden information models to environmental problems in agriculture lies in design of auctions and other bidding mechanisms for paid diversions, conservation set-asides, land retirement, agricultural easements, purchases of development rights, etc., in cases where these programs deal with problems that are local in scope. Land use can be monitored at low cost and land-use restrictions cannot be sidestepped easily, so land-use contracts are likely to be enforceable. Hidden information is likely to be a problem when the number of potential participants is small enough that competition between them is limited.

4. Uncertainty

Agricultural production and its environmental effects are both subject to considerable uncertainty. Two sources of uncertainty can be distinguished (Braden and Segerson 1993): unmeasured heterogeneity and randomness. As noted in the preceding section, both agricultural productivity and environmental effects of agriculture vary according to differences in climate, topography, geology, pest complexes, and other exogenous factors in addition to human variability such as differences in human capital across farmers. When not measured, this heterogeneity constitutes a source of uncertainty. The remaining uncertainty is attributable to randomness, that is, the effects of factors like rainfall that cannot be predicted deterministically.

This section explores three general issues arising due to uncertainty: (1) efficient policy design under uncertainty, (2) regulatory aversion to uncertainty, and (3) the role of information in policy design. We begin by exploring the implications of uncertainty for policy design. We begin by developing a conceptual model of *ex ante* regulation in which government and farmers are risk neutral. We use the model to assess the relative efficiency of incentives (taxes, subsidies, tradable permits) and best practice standards both in general and with respect to specific inputs associated with environmental quality problems. We then turn to implementation in cases where the government is unable to assess emissions of individual farmers. We take up two types of policies: taxes (subsidies) on ambient environmental quality and *ex post* liability for environmental damage. Next, we consider policy design in cases where regulators are averse to uncertainty, either for political reasons or by statutory instruction. Finally, we consider the role of information in policy design. Acquisition of information about states of nature can be considered an input into the production process. It may thus affect the use of inputs that influence environmental quality. Information-intensive farming methods have been widely regarded as opportunities for strengthening stewardship. We discuss briefly their potential effects on environmental quality.

4.1 Taxes, Subsidies, and Best Practice Standards Under Uncertainty

We begin with a situation in which government and/or farmers are risk neutral but must act before uncertainty is resolved, so that they know the results of their actions only stochastically. A modified form of the input-oriented model will be useful for exploring policy design in this context. Let output be a function $f(z,\varepsilon)$ of an input affecting environmental quality z and a random factor ε. Without loss of generality, scale ε so that $f_\varepsilon > 0$, higher realizations of ε are associated with greater output. If $f_{z\varepsilon} < 0$, the input z can be said to be risk-reducing because it increases income more in bad states of nature (low ε) than good states (high ε), thereby reducing the variability of income. Similarly, if $f_{z\varepsilon} > 0$, the input z can be said to be risk-increasing because it increases income more in good states of nature than in bad states, thereby increasing the variability of income.

As is well known from the work of Weitzman (1974), under uncertainty, quantity controls like best practice standards may be preferable to price instruments like Pigouvian taxes, subsidies, or tradable permits. Consider the case where monitoring emissions is excessively costly, so that regulation is applied to inputs associated with environmental quality. Let $\hat{z}$ be the socially optimal usage level of the input z that affects environmental quality and $\tilde{z}$ be the usage level chosen by a farmer facing a price of environmental quality equal to $E\{D_e e_z\}$ that implements the social optimum. Following Weitzman (1974), the difference between the social net benefits under the price incentive and the best practice standard equals, to a second-order approximation around $(\hat{z}, \bar{e})$, where $\bar{e} = E\{e\}$,

$$\begin{aligned}\Delta &\approx \left[pf_z(\hat{z},\bar{\varepsilon}) - v - D_e(e(\hat{z}),\bar{\varepsilon})e_z(\hat{z})\right](\tilde{z} - \hat{z}) \\ &+ \tfrac{1}{2}\left[pf_{zz}(\hat{z},\bar{\varepsilon}) - D_{ee}(e(\hat{z}),\bar{\varepsilon})e_z^2(\hat{z}) - D_e(e(\hat{z}),\bar{\varepsilon})e_{zz}(\hat{z})\right](\tilde{z} - \hat{z})^2 \\ &+ \left[pf_{z\varepsilon}(\hat{z},\bar{\varepsilon}) - D_{e\varepsilon}(e(\hat{z}),\bar{\varepsilon})e_z(\hat{z})\right](\tilde{z} - \hat{z})(\varepsilon - \bar{\varepsilon}).\end{aligned}$$

If the farmer is risk-neutral, the optimal usage level of the input z satisfies

$$E\{pf_z - v - E\{D_e e_z\}\} = 0.$$

A first-order approximation of this necessary condition around $(\hat{z}, \bar{e})$ indicates that

$$\tilde{z} - \hat{z} \approx pf_z(\hat{z},\bar{e}) - v - E\{D_e e_z\} - \frac{f_{ze}(\hat{z},\bar{e})}{f_{zz}(\hat{z},\bar{e})}(e - \bar{e}).$$

Substituting for $(\tilde{z} - \hat{z})$ in Δ and taking expectations gives

$$E\{\Delta\} \approx s^2 \left\{ \frac{D_{ze} e_z f_{ze}}{f_{zz}} - \frac{f_{ze}^2}{2 f_{zz}^2}\left[pf_{zz} + (D_{ee} e_z^2 + D_e e_{zz})\right] \right\}.$$

The second term in curly brackets corresponds to Weitzman's main result (p. 484), derived under an assumption of no correlation between the random elements affecting benefits and costs. If environmental damage is more sensitive to random variations than crop productivity, the term in square brackets is positive and the effect on $E\{\Delta\}$ is negative, that is, the social loss from best practice standards is less than the social loss from pricing environmental quality. If crop productivity is more sensitive to randomness than environmental damage, on the other hand, then incentives are likely to be preferable.

The first term in curly brackets represents the effect of correlation between random variations in crop productivity and environmental damage, and corresponds to the finding noted in Weitzman's footnote 1 (p. 485) and in Stavins (1996). Its sign depends on the combined risk effects of z on crop productivity and environmental damage. Recall that ε is scaled so that $f_\varepsilon > 0$, that is, higher values of ε represent better (random) growing conditions. The effect of growing conditions on environmental damage likely depends on the type of damage, although materials balance considerations suggest that environmental damage is likely lower under better growing conditions, that is, $D_\varepsilon < 0$ and $D_{z\varepsilon} < 0$. For example, one would expect crop nutrient uptake to be greater under better growing conditions and nutrient runoff and/or leaching correspondingly less, implying $D_{z\varepsilon} < 0$. Similarly, herbicide uptake by weeds is likely greater (and thus residual herbicide in soils likely less) when growing conditions are better and thus weed densities are greater. If $D_{z\varepsilon} < 0$, the sign of this first term depends on the risk effect of z on crop productivity. If the input z is risk-increasing in terms of crop productivity, then incentives are likely to be preferable. If, on the other hand, the input z is risk-reducing in terms of crop productivity, then best practice standards are likely to be preferable. More generally, incentives are more likely to be preferable when $f_{z\varepsilon}$ and $D_{z\varepsilon}$ have opposite signs, while best practice standards are more likely to be preferable when $f_{z\varepsilon}$ and $D_{z\varepsilon}$ have the same signs.

4.1.2 Incentives versus Best Practice Standards in Agriculture

Weitzman's main result is that the choice between incentives and best practice standards depends on the relative sensitivity of crop productivity and environmental damage to uncertainty. The empirical economic literature has not addressed this issue at all. However, when $D_{z\varepsilon} > 0$, as generally expected, Weitzman's argument also suggests that economic incentives are more likely to be preferable under uncertainty for risk-increasing inputs while best practice standards are more likely to be preferable for risk-reducing inputs. In this section, we briefly review the literature on risk effects of the agricultural inputs associated with environmental quality. Additional treatment of this issue can be found in Moschini and Hennessy elsewhere in this Handbook.

Fertilizers. Fertilizers are widely believed to be risk-increasing, at least in rainfed agriculture. When soil moisture is low (and thus growing conditions are poor), crop uptake of macronutrients (nitrogen phosphorus, potassium) and thus marginal nutrient productivity are low. Nitrogen in particular may cause crop burn and thus have negative marginal productivity when soil moisture is low. As growing conditions improve, crop uptake and the marginal productivity of nutrients increase. In other words, nutrient productivity and states of nature tend to be positively correlated, $f_{z\varepsilon} > 0$.

Nitrogen has been the most extensively studied macronutrient. Most empirical studies have found it to be risk-increasing. Roumasset et al. (1989) survey the literature on fertilizer and yield variability through the mid-1980s, with an emphasis on developing-country applications. Of the roughly 20 studies they cite, all but two (Farnsworth and Moffitt 1981, and Rosegrant and Roumasset 1985) found nitrogen to be risk-increasing. In all cases, however, the estimated risk effects of nitrogen were small. The subsequent literature contains a similar pattern of results. Studies using estimated parameters of a conditional beta distribution for Iowa corn yields found that nitrogen increased the variance of yield (Nelson and Preckel 1989; Love and Buccola 1991; Babcock and Hennessy 1996). Horowitz and Lichtenberg (1993) found that corn growers in the U.S. Corn Belt who purchased crop insurance used more nitrogen per acre, indicating that nitrogen is risk-increasing.

There is less evidence about risk effects of other macronutrients. Studies of Iowa corn indicated that phosphorus is risk-increasing while potassium is risk-decreasing (Nelson and Preckel 1989; Love and Buccola 1991), while Horowitz and Lichtenberg (1993) found that corn growers who purchased crop insurance did not use phosphorus or potassium in significantly different per acre amounts than those who did not, suggesting that whatever risk effects these chemicals have are likely small. Finally, Smith and Goodwin (1996) found that Kansas wheat growers who purchased crop insurance spent less on fertilizers in the aggregate, a category that includes items thought to have risk effects of opposite signs.

Soil erosion. As noted earlier, preserving the value of their land gives farmers an incentive to engage in soil conservation by presenting farmers with a tradeoff between current yield and the future value of their land. Their choice of erosion level (equivalently, soil conservation effort) is simultaneously a choice about investment in land quality. Both current yield and future land prices are subject to uncertainty. Ardila and Innes (1993) note that the effect of increased uncertainty (in the form of a Sandmo-type mean-preserving spread) on optimal soil conservation depends on the relative sizes of yield and land price risks. We are unaware of any empirical studies on the relative sizes of these risks and are thus unable to draw inferences about appropriate policy design.

Pesticides. The literature on risk effects of pesticides is somewhat confusing, in part because it seems to have arisen from a misunderstanding regarding terminology between economists and crop scientists. The latter noted that farmers frequently used pesticides preventively, that is, before the degree of infestation was observed. They argued on that basis that farmers were responding to the risk of infestation rather than actual infestation and thus characterized pesticide use in terms of insurance (van den Bosch and Stern 1962). Economists responded to this characterization by developing models in which pesticides reduced the variability of income or output, producing the conventional wisdom that pesticides are risk-reducing. Feder (1979), for example, analyzed pesticides using a Sandmo-type model in which pest damage was assumed to be additively separable from potential output and equal to the product of pest population size, damage per pest, and survivorship from pesticide application. He found that pesticides are risk-reducing when there is uncertainty about pest population size or damage per pest, but risk-increasing when there is uncertainty about the marginal effectiveness of the pesticide.

Farmers' use of preventive treatments is not, of course, in and of itself evidence of risk aversion. Even when pest infestations are observable, preventive treatment may be more profitable on average if rescue treatment is insufficiently effective, if the error in estimating infestation from observed levels is excessive, or if monitoring (scouting) is sufficiently costly. Moreover, correlation between random factors affecting price or potential yield and pest damage may create situations in which pesticides increase rather than decrease yield and income variability (Pannell 1991; Horowitz and Lichtenberg 1994). For example, greater early-season soil moisture and solar radiation may promote the growth of all plants, crops and weeds included, while low moisture and solar radiation may lead to poor plant growth. In such situations, the marginal productivity of herbicides is likely to be higher in good states of nature than bad ones, implying that pesticides are risk-increasing. Similarly, insecticides are thought to reduce the stability of crop ecosystems during the growing season by suppressing invertebrate predators and by altering competition among insect, weed, and disease species in ways that create less tractable pest problems (see for example Bottrell 1979).

The empirical evidence available to date suggests that pesticides frequently are risk-increasing. Using a Just-Pope production function, Farnsworth and Moffitt (1981) found that pesticides increased the yield variability of irrigated cotton in California's San Joaquin Valley. A more recent study of the same crop and region by Hurd (1994) using a similar specification of production found that pesticides had no impact on yield variability. Antle (1988), using the moment-based econometric procedure of Antle (1983), found that insecticide use had no statistically significant effect on yield variability of processing tomatoes in California. Gotsch and Regev (1996) and Regev, Gotsch, and Rieder (1997), also using the moment-based approach of Antle (1983), found that fungicide use increased the variability of wheat revenue (which captures both quantity and quality effects) in Switzerland. Horowitz and Lichtenberg (1993) found that corn growers in the U.S. Corn Belt who purchased crop insurance applied more herbicides and insecticides and spent more on pesticides, suggesting that pesticides are risk-increasing.

In cases where pesticides are risk-increasing, economic incentives are likely to be preferable to best practice standards. In such cases, shifting to incentive-based pesticide regulation could improve performance with respect to uncertainty as well as with respect to heterogeneity. Overall, however, the risk effects of pesticides have not been studied sufficiently to permit broad generalizations, and it is possible that broad generalizations may never be possible. Herbicides, insecticides, and fungicides may differ in their risk effects, and those risk effects may differ across crops and regions.

4.1.2 Moral Hazard

Environmental quality is frequently influenced by inputs that are neither traded in markets nor readily observable. Examples include the observance of setback requirements and similar wellhead protection measures in mixing and loading pesticides; banded application of pesticides; scouting prior to pesticide application; and split application of fertilizers as a means of increasing the share of nutrients taken up by crops. Input taxes and subsidies are ineffective in such cases. When pollution control effort of these kinds can neither be observed directly nor inferred from production, the regulatory problem is characterized by moral hazard, or hidden action.

Nevertheless, a social optimum can be attained provided that ambient environmental quality is observable and that all farmers influencing environmental quality face the same set of random factors. In such cases, observing ambient environmental quality is equivalent to observing each farmer's pollution control effort, and moral hazard is not a factor. Holmstrom (1982) shows that a principal contracting for production of a single output produced collectively by risk-averse agents can induce those agents to exercise first-best effort as long as the reward scheme is not required to be budget-balancing. Rasmusen (1987) shows that, if agents are risk-averse, it is possible to attain a social optimum with mechanisms that are budget-balancing by random differential treatment of one out of many agents. Segerson (1988) applies Holmstrom's result to a nonpoint-source pollution control problem in which (crop) output is nonrandom while environmental quality is stochastic and influenced by unobservable effort. The optimal mechanism in this context consists of two parts. If actual observed ambient pollution exceeds a pre-specified level, each farmer pays a pollution tax proportional to the difference between actual and that pre-specified level. If actual observed ambient pollution is less than the pre-specified level, each farmer receives a pollution subsidy proportional to the difference between actual and the pre-specified level. The socially optimal level of crop production can be attained in the long run by adding to the pollution tax a lump-sum payment or tax as needed to induce appropriate entry or exit. This result generalizes to the case where both crop output and environmental quality are random and farmers are risk-averse. As long as environmental quality consists of observable ambient pollution from uniformly mixed emissions, a combination of crop insurance and an ambient pollution tax will achieve a first-best allocation of crop output and environmental quality (Chambers and Quiggin 1996). We examine interactions between insurance and environmental policy at greater length in Section 5.2 below.

If there are many farmers with differential impacts on ambient environmental quality, ambient pollution taxes can be used to achieve first-best environmental quality on average if the regulator has perfect information about the effects of each farmer's environmental quality effort on the probability distribution of ambient pollution (Segerson 1988), including fate and transport of pollutants (Cabe and Herriges 1992). This information is costly to obtain. Xepapadeas (1995) shows there may be conditions under which farmers can be induced to reveal fully their own environmental quality effort. Farmers must be averse to the riskiness of tax payments (but not crop production or income) and able to reduce their own tax liability under an ambient pollution tax by reporting indicators of environmental quality effort. Participating in integrated pest management (IPM) programs, obtaining conservation plans from government agencies, and applying for cost sharing to install runoff or erosion control facilities might be interpreted as methods of reporting such effort.

Ambient pollution taxes capable of achieving first-best environmental quality may be too high to be acceptable politically. For example, it may be optimal to charge each farmer a tax equal to the marginal damage from all ambient pollution in order to create incentives for first-best environmental quality effort (Segerson 1988).

4.1.3 Environmental Quality and the Tort System

A number of authors have argued that *ex post* liability for harm can provide sufficient incentives to ensure that farmers exercise socially optimal precautions against stochastic environmental damage. There has been particular interest in the role of the tort system in mitigating groundwater quality degradation, since well owners can take legal action in the event of impairment of drinking well water quality. Segerson (1990) considers the extent to which farmers and pesticide manufacturers should be held liable for pesticide contamination of drinking water wells. Wetzstein and Centner (1992) examine the potential effects of legislation that would replace strict liability for groundwater contamination by agricultural chemicals with a negligence standard.

Menell (1991) has argued forcefully that the tort system gives polluters few incentives to take action to mitigate environmental damage because of the difficulty of establishing causal links between (1) the actions of any single agent and ambient pollutant concentrations and (2) ambient pollutant concentrations and harm suffered. The source or sources of well water contaminants, for instance, cannot be readily identified.

The current state of biomedical knowledge is insufficient to prove to a legal standard that the low concentrations of pesticides or nitrate typically found in drinking water cause long-term health effects or other forms of environmental damage. Most such health effects have multiple causes, making identification of causality difficult, even in the case of acute health effects from direct exposure to pesticides. Moreover, Menell argues, the tort system is extremely costly, time-consuming, and inequitable. Furthermore, inadequate financial resources may prevent those harmed from bringing suit, further weakening incentives for precautionary behavior (see for example Shavell 1987).

Davis, Caswell, and Harper (1992) compare tort liability with direct regulation and workers' compensation as means of protecting farm workers from pesticide poisonings. Their simulation analysis indicates that experience-rated workers' compensation is the most cost-effective of these policy instruments. The fact that insurance premiums increase as the number of valid claims filed goes up regardless of the specific cause of those claims gives farmers strong incentives to take precautionary action. Tort liability is less cost-effective because farm workers frequently fail to seek medical care, because doctors frequently fail to attribute observed symptoms to pesticides, and because farm workers lack resources to support legal action, while administrative regulation has high enforcement costs.

4.2 Regulatory Aversion to Uncertainty

Thus far, the government has been treated as neutral with respect to uncertainty, that is, interested only in environmental quality on average. Indeed, government is typically assumed to be risk-neutral because its large size permits adequate diversification. But this assumption may not be appropriate for environmental issues.

Environmental quality tends to be idiosyncratic and thus non-diversifiable. It is difficult, for example, to imagine means of compensating for an increment in premature deaths due to exposure to environmental toxicants or for the destruction of rare ecosystems. Governments are frequently sensitive to the prospect of making mistakes, especially to the possibility that realized environmental damage will turn out to be worse than expected. For example, most environmental legislation in the United States requires regulation to incorporate a margin of safety adequate to protect against uncertainty in meeting environmental quality standards. It is thus reasonable to treat governments as averse to uncertainty about environmental quality.

Lichtenberg and Zilberman (1988) have studied the engineering approach to uncertainty management implicit in the requirement of an adequate margin of safety. They argue that this requirement corresponds to a safety-fixed decision problem in which regulators choose instruments to minimize the social cost of regulation subject to the constraint that a nominal standard not be exceeded with more than a specified frequency. Beavis and Walker (1983) use a similar framework to derive optimal effluent taxes on random discharges from a set of polluters. Bigman (1996) proposes a modification of this criterion that incorporates the size of the deviation from the nominal standard.

In terms of the modified output-based model of the preceding section, optimal regulation in Lichtenberg and Zilberman's (1988) analysis involves choosing inputs x and environmental quality-enhancing effort a to minimize cost wx + ra subject to the technological constraint $(y,q,x,a) \in T$ and the safety-fixed constraint $\Pr\{q > \overline{q}\} \leq 1-\alpha$ (alternatively, $\Pr\{q \leq \overline{q}\} \geq \alpha$), where α, the frequency with which the nominal standard $\overline{q}$ is met, corresponds to the statutory margin of safety. As Lichtenberg, Zilberman, and

Bogen (1989) note, this decision problem generates an uncertainty-adjusted cost function $C(y,q,\alpha) = \min\{wx + ra: (y,q,x,a) \in T, \Pr\{q \leq \overline{q}\} \geq \alpha\}$. The margin of safety α can be interpreted as a measure of society's aversion to uncertainty. A higher value of α corresponds to lower tolerance for violations of the nominal standard and thus greater aversion to uncertainty. This interpretation suggests that social benefits, too, should reflect social preferences for uncertainty about environmental quality, so that the social benefit function should be written as $U(y,q,\alpha)$ and maximized by choices of crop output, environmental quality, and margin of safety.

Lichtenberg and Zilberman (1988) examine theoretically the properties of cost-minimizing regulatory decisions in the case where the safety-rule constraint can be represented as a weighted sum of the mean and standard deviation of environmental quality. They show that the optimal regulatory policy consists of a portfolio of instruments of which some have comparative advantage in enhancing environmental quality on average while others have comparative advantage in reducing uncertainty about environmental quality. They show that increased aversion to uncertainty (a higher margin of safety α) leads to a higher total cost of regulation and increased use of instruments with comparative advantage in reducing uncertainty, a class that includes research and data acquisition. The marginal cost of environmental quality will fall, however, and use of instruments with comparative advantage in enhancing environmental quality on average may fall as well. When there is greater background uncertainty about environmental quality, it becomes efficient to increase reliance on instruments with comparative advantage in enhancing environmental quality on average, while the use of instruments with comparative advantage in reducing uncertainty may fall.

Empirical applications of this framework include Lichtenberg, Zilberman, and Bogen (1989), who estimate the cost of reducing the risk of cancer from pesticide contamination of drinking water; Lichtenberg and Zilberman (1989), who estimate the cost of mitigating the risk of gastroenteritis from consumption of shellfish contaminated by dairy wastes; Hanemann, Lichtenberg, and Zilberman (1989), who estimate the cost of meeting standards for selenium in river water; Harper and Zilberman (1992), who estimate the cost of reducing farm workers' cancer risk from insecticide exposure; and Lichtenberg and Penn (1998), who estimate the cost of meeting nitrate standards in well water. Estimated uncertainty premia (that is, incremental costs due to increases in the margin of safety) range from 1 percent or less in cases of lax standards and low margins of safety to as much as 35 percent for stringent standards and high margins of safety.

4.3 Information Acquisition: IPM and Precision Agriculture

As noted previously, uncertainty springs from two types of sources, randomness and unobserved or unmeasured heterogeneity. Uncertainty from both sources can be reduced or, in some cases, eliminated entirely. Weather forecasts can be used to reduce uncertainty about rainfall, which affects optimal fertilizer application rates (Babcock 1990), or uncertainty about evapotranspiration demand, which influences optimal irrigation water application rates (Cohen et al. 1998). Scouting can reduce uncertainty about crop disease (Carlson 1970) or insect pressure (Moffitt 1986; Stefanou, Mangel, and Wilen 1986) and thus influence optimal pesticide use. Soil tests can reduce or eliminate uncertainty about soil fertility, water infiltration rates, or other soil characteristics due to unmeasured soil heterogeneity (Babcock and Blackmer 1992; Babcock, Carriquiry, and Stern 1996). In some cases it may be feasible to wait until random events are realized and act with certainty in accordance with the realization of the random variable. Scouting falls into the latter category if it is sufficiently accurate, since it is generally undertaken with the intent of postponing the use of pesticides or other pest control methods until the extent of infestation has been observed.

Information-intensive technologies are widely cited in policy discussions as potential means of improving environmental quality and farm profitability simultaneously, that is, of enhancing stewardship. This proposition has received relatively little careful economic analysis. The few analyses that have been performed suggest a need for caution in making such inferences.

Reducing uncertainty by acquiring information can affect environmental quality in two major ways. First, it can lead to changes in the use of inputs that impair environmental quality (equivalently, in the use of inputs that enhance environmental quality). Second, it can improve targeting of inputs to improve application efficiency, as was discussed in Section 2.2.2. In what follows, we discuss potential environmental quality effects in both of these cases.

4.3.1 Information, Input Use, and Environmental Quality

Reductions in uncertainty due either to measurement of previously unobserved heterogeneity or to more accurate forecasts of random events have been modeled as mean-preserving contractions of the distribution of the unobserved factor or by using explicit Bayesian models. Most studies have examined the acquisition of information using single-input production models, such as the model in which output $y = f(y,\varepsilon)$, used in the preceding sections. In the present case, the random factor ε represents either unobserved but measurable factors of production such as soil nutrient stocks or the water infiltration rate of the soil at a specific location, or random factors such as pest infestation rates. Feder's (1979) study of the risk effects of pesticides, discussed above, is undertaken with scouting explicitly in mind. Babcock and Blackmer (1992) model pre-sidedress nitrogen test information as a mean-preserving contraction in the distribution of the soil nitrogen stock in a model in which nitrogen fertilizer and soil nitrogen are treated as perfect substitutes. Feinerman, Letey, and Vaux (1983) use this framework to investigate the effects of uniformity of soil infiltration rates on optimal irrigation water application rates. (See also Moschini and Hennessy elsewhere in this Handbook for a more general discussion.)

Reductions in uncertainty influence input use even if farmers are risk-neutral, as can be seen using a second-order approximation of expected profit around the mean of ε

$$E\{pf(z,\varepsilon)\} \approx p\left[f(z,\bar{\varepsilon}) + \tfrac{1}{2} f_{\varepsilon\varepsilon}(z,\bar{\varepsilon})\sigma^2\right] - vz .$$

The term $\frac{1}{2}f_{\varepsilon\varepsilon}\sigma^2$ is what Babcock and Shogren (1995) term production risk. It is negative by the concavity of the production function in ε, i.e., $f_{\varepsilon\varepsilon} < 0$. The first-order condition defining optimal use of the input z is

$$p\left[f_z(z,\bar{e}) + \tfrac{1}{2} f_{eez}(z,\bar{e})s^{2}\right] - v = 0 .$$

A reduction in uncertainty (i.e., in σ^2) leads to decreased (increased) use of z if $f_{\varepsilon\varepsilon z}$ > (<) 0, that is, if increases in z reduce (increase) the concavity of the production function in ε. Alternatively, a reduction in uncertainty leads to decreased (increased) use of z if increases in ε make the input more (less) risk-increasing, that is, if $\partial f_{\varepsilon z}/\partial\varepsilon$ > (<) 0. There is no extant empirical evidence about the sign of this third derivative, however.

Information has value to farmers even if they are risk-neutral. By the envelope theorem, a reduction in σ^2 increases the farmer's expected profit by the amount $\frac{1}{2}pf_{\varepsilon\varepsilon}$.

But acquiring information about ε may not increase overall social welfare. If environmental damage is not taken into account in farmers' input use decisions, the social value of information acquisition is

$$½\left[pf_{ee} - D_{ee}\right] - ½s^{2}\left[D_{e}e_{z} + D_{eez}\right]\frac{\partial z}{\partial s^{2}}.$$

The first term in square brackets is negative, but the second term can be positive. In the absence of regulation, then, reductions in uncertainty can decrease social welfare by exacerbating environmental quality problems.

There have been few, if any, studies investigating optimal sampling or testing strategies in an economic context, that is, in the context of improved crop production decisions. Carlson (1970) estimates the impact of peach disease-loss forecasts on the mean and standard deviation of returns from four fungicide use strategies. Stefanou, Mangel, and Wilen (1986) derive scouting-based spraying strategies for lygus bug on cotton and estimate the value of scouting information of varying accuracy. Babcock, Carriquiry, and Stern (1996) present a Bayesian model of soil nitrogen testing and an application to Iowa corn production. There have also been discussions of scouting methods in the entomology and weed science literatures, but they have not been combined with economic models of pest management decision making. Grid sample size for soil testing has emerged recently as an important issue given the costliness of sampling (see for example National Research Council 1997). One issue of interest is whether decreased grid size improves the accuracy of information about important soil characteristics. There is some evidence that variability of soil characteristics may not decrease appreciably at smaller sampling scales (National Research Council 1997). If it does not, then soil sampling may have limited effects on environmental quality.

4.3.2 Information and Application Efficiency

Information can also be used to target input application more precisely, thereby improving environmental quality. For example, scouting may reduce risks to human health and wildlife from pesticide use by reducing pesticide applications on average (although not necessarily in every single year), by permitting the use of narrower-spectrum chemicals with fewer spillover effects, or by permitting spot treatments of areas with high infestation rates. Similarly, soil testing allows farmers to take existing soil nutrient stocks into account in choosing chemical fertilizer application rates, reducing excess applications and thus eventual leaching. Information on expected crop evapotranspiration rates derived from weather forecasts allows farmers to match irrigation application rates with crop demands. In most cases, optimal water use declines, creating the potential to improve instream environmental quality by reducing water diversion for irrigation (Cohen et al. 1998). However, it is not necessarily true that improved information will reduce the use of inputs that impair environmental quality, even on average and even on only the intensive margin.

The following analysis, adapted from Moffitt (1986, 1988), uses a modified form of the input-oriented model to discuss this point in the context of scouting. Let agricultural output be a function $f(z,\varepsilon)$ of an input affecting environmental quality z and a random factor ε such that $f_\varepsilon > 0$. Let $z^*(\varepsilon)$ be the optimal pesticide application rate when pest pressure is ε, defined by

$$pf_z(z^*(\varepsilon),\varepsilon) - v = 0.$$

Assume also that there exists a threshold level of pest pressure ε^c such that $z^*(\varepsilon) = 0$ for $\varepsilon \leq \varepsilon^c$ (Headley 1972; Mumford and Norton 1984; Moffitt 1988).

The optimal pesticide application rate of a risk-neutral farmer engaging in preventive treatment z^p is defined by

$$\int pf_z(z^p, \varepsilon)\psi(\varepsilon)d\varepsilon - v = 0,$$

where $\psi(\varepsilon)$ is the probability density of ε and $\Psi(\varepsilon)$ is the corresponding cumulative distribution. There exists a level of pest pressure ε^p such that

$$pf_z(z^p, \varepsilon^p) - v = \int pf_z(z^p, \varepsilon)\psi(\varepsilon)d\varepsilon - v = 0,$$

that is, $z^p = z^*(\varepsilon^p)$. The difference in pesticide use between preventive treatment and the scouting regime is

$$z^p\Psi(\varepsilon^c) + \int_{\varepsilon^c}^{\varepsilon^p}\left[z^p - z^*(\varepsilon)\right]\psi(\varepsilon)d\varepsilon + \int_{\varepsilon^p}\left[z^p - z^*(\varepsilon)\right]\psi(\varepsilon)d\varepsilon.$$

If $f_{z\varepsilon} > 0$, $z^*(\varepsilon) > (<)\ z^p$ when $\varepsilon < (>)\varepsilon^p$. The first and third terms of this difference are thus positive, while the second term is negative. If $\Psi(\varepsilon^p)$-$\Psi(\varepsilon^c)$ is sufficiently large, total pesticide application will be greater with scouting than under a preventive regime. (If $f_{z\varepsilon} < 0$, the inequalities will be reversed but the basic analysis still holds.) In such cases, providing free scouting services could have detrimental effects on environmental quality and social welfare overall.

Constraints on timing of application may be a significant impediment to the use of such information-based application strategies. Feinerman, Choi, and Johnson (1990) model split application of nitrogen fertilizer as a form of increased application efficiency. They assume that initial soil nitrogen stocks and pre-plant applications of nitrogen are subject to leaching, runoff, and volatilization losses, while side-dress applications are not. Let z_0 denote the initial stock of soil nitrogen, z_1 the pre-plant application rate, z_2 the side-dress application rate, and (1-h) denote losses of pre-plant nitrogen, all of which are known with certainty. Effective nitrogen s(z,h) is then $z_2+h[z_0+z_1]$. Side-dressing is more efficient, but some pre-plant application may be optimal because of the risk that adverse weather conditions (too much rainfall) will prevent side-dressing. Let ε represent soil moisture, scaled to lie in the unit interval, so that crop production is $f(z_2+h[z_0+z_1],\varepsilon)$. Environmental damage is $D(e(1-h)[z_0+z_1],\varepsilon)$. Side-dress application is infeasible when soil is excessively wet, that is, $z_1 = 0$ when $\varepsilon \geq \varepsilon^c$, making pre-plant application optimal (thereby creating environmental damage) for both risk-neutral and risk-averse farmers when nitrogen is sufficiently cheap. Lichtenberg, Spear, and Zilberman (1994) note that re-entry regulation of pesticides may create similar disincentives for using reactive rather than preventive pesticide application. In such cases, even subsidies for information acquisition may be of limited effectiveness.

Matching input application rates with crop ecosystem uptake rates requires knowledge about both natural characteristics affecting production and about the ways in which applied inputs interact with those characteristics, that is, about technological structure. More precise measurement of field conditions has little value without knowledge of the structure of crop production technologies (National Research Council 1997). Biological knowledge may be an important source of *a priori* information permitting improved specification of agricultural technology structure.

Modeling crop nutrient response provides one important example. A series of papers by Paris and his colleagues and by others examines the applicability of the von Liebig hypothesis of limitationality to nutrient response modeling. This hypothesis has two major features: (1) limited substitutability between nutrients and (2) yield plateaus, that is, zero marginal productivity of non-limiting nutrients at some levels. Empirical studies on corn provide evidence of this limitationality: Best-fitting models combine yield plateaus and diminishing marginal nutrient productivity (see for example Lanzer and Paris 1981; Ackello-Ogutu, Paris, and Williams 1985; Grimm, Paris, and Williams 1987; Paris and Knapp 1989; Frank, Beatty, and Embleton 1990; Cerrato and Blackmer 1990; Paris 1992; Chambers and Lichtenberg 1996). Limitationality could have important implications for understanding the effects of policy-induced changes in fertilizer use on nutrient pollution. First, fertilizer recommendations generated from polynomial specifications tend to exceed those from a von Liebig specification. If the von Liebig specification is correct, standard functional forms generate excessive nutrient application rates (Ackello-Ogutu, Paris, and Williams 1985). Second, policies for enhancing water quality are typically targeted toward the limiting nutrient governing eutrophication (usually either nitrogen or phosphorus). But reductions in application rates of one nutrient could make it limiting in crop production, leading to increased runoff of other nutrients. For example, application of poultry litter to crops has been advocated as a means of reducing nitrogen loadings into the Chesapeake Bay. Application rates are targeted on crop nitrogen requirements, leading to accumulations of excess phosphorus. It now appears possible that excess phosphorus may be leaching from fields, exacerbating the Bay's nutrient pollution problem.

In a similar vein, Lichtenberg and Zilberman (1986a) attempt to introduce crop ecosystem thinking into models of pesticide productivity. They argue that pesticides should be treated differently from normal inputs because they limit damage rather than contribute to potential output. They propose a model in which pesticides (and other inputs) produce an intermediate input called abatement. The fact that abatement cannot exceed potential output implies that the marginal product of pesticides and other damage control inputs declines faster than most first-order approximations such as a log-linear model. Babcock, Lichtenberg, and Zilberman's (1992) analysis of apple production found a substantial difference in marginal productivity and thus in profit-maximizing pesticide application rate recommendations. The appropriateness of this damage control model has been the subject of some debate. Several studies have found that some form of abatement function provides a better model fit than a standard log-linear model (Carrasco-Tauber and Moffitt 1992; Babcock, Lichtenberg, and Zilberman 1992; Chambers and Lichtenberg 1994), while others have found generic functional forms better fitting (Crissman, Cole, and Carpio 1994; Carpentier and Weaver 1997). (Carpentier and Weaver's claim is weakened by the fact that they compared a generic specification with fixed farmer effects with a damage control specification without those fixed effects.)

An additional attraction from a policy perspective is the fact that the damage control model permits inference of crop damage, that is, of the percentage of crop lost to pests under any configuration of pest control inputs. The notion that relative crop losses to pests have remained virtually constant at about 30 percent over the past three decades (Pimentel et al. 1991) has been one of the main arguments used to motivate the need for stricter overall regulation of pesticides. The estimates on which this claim is based are derived from a potpourri of field studies. Chambers and Lichtenberg (1994) used a dual formulation of the damage control model to estimate the share of aggregate U.S. crops lost, and found that losses were much smaller initially and have declined markedly due to pesticide use. They also found no evidence of a "pesticide treadmill" in which ecological damage leads to a spiral of ever-increasing pesticide use.

5. Interactions Among Agricultural, Resource, and Environmental Policies

Agriculture has been treated thus far as a competitive industry characterized by market failures in the provision of environmental quality. In most countries, however, agricultural markets are also subject to significant distortions, primarily due to government intervention. Virtually all countries have agricultural policies that influence agricultural markets. In developed countries, such policies are typically designed to bolster farm income and/or stabilize the prices of agricultural commodities. In developing countries, such policies are typically designed to reduce food prices in urban areas and transfer income from agriculture to other sectors of the economy (see for example Schiff and ValdJs 1992). Many countries also have policies designed to promote the use of inputs associated with environmental quality such as chemical fertilizers, pesticides, water, and land. Some such policies have survived in developed countries, for example, irrigation water subsidies originally introduced in order to promote economic growth in the western United States.

As a result, it is necessary to design and evaluate environmental policies aimed at agriculture in a second-best context that takes into account the distortions introduced by agricultural and other resource policies (Lichtenberg and Zilberman 1986b). These distortions can be substantial. Lichtenberg and Zilberman (1986b), for example, analyze the impacts of a hypothetical marginal output cost-increasing environmental regulation affecting a crop subject to a simple deficiency payment program. The supply price farmers face exceeds the demand price, resulting in excessive output and deadweight efficiency loss. An environmental regulation that increases the marginal cost of production reduces producer and consumer surplus in the affected market but reduces the size of the deadweight loss as well. Back-of-the-envelope calculations using supply and demand elasticities, output levels, market prices, and target prices typical of the mid-1980s indicate that the reduction in deadweight loss largely counteracts reductions in market consumer and producer surplus for crops like cotton and rice. Moreover, estimates of market-level economic costs of regulation calculated under an assumption that markets are perfectly competitive far exceed the true market-level costs.

This section discusses the implications of interactions between environmental and agricultural policies for policy design. We focus on three kinds of agriculture sector policies: (1) those designed to raise average income (price and income support programs), (2) those designed to reduce income variability (price stabilization and crop insurance), and (3) those designed to promote certain kinds of agricultural production technologies by subsidizing key inputs such as water, fertilizers, or pesticides.

5.1 Agricultural Price and Income Policies

Agricultural price and income policies are used widely. In developed countries, the main goal tends to be maintaining the farm sector by increasing farm income. In developing countries, these policies are typically aimed at transferring income from the farm sector to urban consumers and/or industries (Schiff and ValdJs 1992). These policies typically drive a wedge between the price farmers face and consumers' willingness to pay for agricultural output. In the context of the output-based model introduced earlier, farmers' decision problem in such situations is to choose agricultural output y and environmental quality q to maximize profit $py - C(y,q)$, where p is determined by government agricultural policy. The necessary conditions are

$$p - C_y(y,q) = 0$$

$$C_q(y,q) = 0.$$

If government policy raises the supply price p above consumers' willingness to pay $U_y(y,q)$, agricultural output exceeds the socially optimal level. If agricultural output and environmental quality are substitutes ($C_{yq} < 0$), such a price support policy exacerbates environmental quality problems. If agricultural output and environmental quality are complements ($C_{yq} > 0$), then price support mitigates environmental quality problems.

Environmental quality and agricultural output are likely to be substitutes in situations where environmental problems are associated with the use of agricultural chemicals (fertilizers, pesticides) or with conversion of land to agricultural uses (deforestation, loss of wetlands, wind erosion from cultivation of virgin prairie). In these cases, price support programs like those used widely in developed countries are likely to worsen environmental quality problems. In developing countries, however, agricultural policies typically depress farm-level prices and are thus likely to mitigate environmental damage.

Environmental quality and agricultural output are clearly complements in cases where agriculture provides environmental quality directly, as in the case of scenic amenities from farming. Price support programs in developed countries thus tend to promote farmland preservation and thus the provision of such amenities, at least to the extent that price support policies apply to the products typically raised near urban areas.

The case of soil erosion is more complex. Price supports increase the return to current production and thus the return to erosion. At the same time, they tend to increase expected future prices and thus the marginal user cost of soil erosion, that is, the expected returns to soil conservation. Consider the input-based model used earlier to discuss soil conservation at the farm level in the case where the price of agricultural output p is determined by government policy. An increase in p above the competitive market equilibrium level changes the soil erosion rate z by

$$\frac{\partial z}{\partial p} = -\frac{R_{pz} + L_{ep} e_z}{R_{zz} + L_{ee} e_z^2 + L_e e_{zz}}.$$

The sign of $\partial z/\partial p$ equals the sign of the numerator. The term R_{pz} represents the effect of an output price increase on current returns to erosion. It is positive if erosion is a normal input. The term $L_{ep}e_z$ represents the change in the future value of the farm L(p,e(z)) due to increased soil losses e_z. One would expect an increase in p to increase the marginal value of soil, i.e., $L_{ep} < 0$. Thus, the effect of price support on soil erosion is ambiguous, as LaFrance (1992) discusses in detail. Claims to the contrary depend on specialized assumptions about the tradeoff between present and future income. For example, Barrett's (1991) claim that soil conservation is invariant with respect to price policy depends on the rather unrealistic assumptions of stationary prices and additive separability between soil depth/soil erosion and the use of other variable inputs (LaFrance 1992). Clarke (1992) uses a model in which erosion *per se* does not affect crop productivity in deriving the result that price supports unambiguously reduce erosion.

Price support programs are complex. Most require farmers to comply with production controls like acreage set-asides and/or meet eligibility conditions like base acreage requirements in order to limit budgetary exposure. As a result, these programs generate countervailing incentives. Moreover, most environmental quality problems are regional or local in scope and must thus be analyzed on a regional or local scale. Aggregate nationwide assessments are relatively uninformative.

If agriculture is characterized by constant returns to scale, as is widely believed for crop production, price and income support programs may influence the use of inputs that affect environmental quality on both the intensive and extensive margins, as discussed in Section 4. Rausser, Zilberman, and Just (1984) discuss the effects of deficiency payment programs with set-asides on land allocations using such a model. They argue that farmers may find it profitable to rent low-quality land in order to meet set-aside requirements, i.e., that set-asides may create a market for "diversion-quality" land. Just and Antle (1989) discuss extensive-margin effects of alternative policy configurations in greater detail.

If agricultural production is not characterized by constant returns to scale, price and income support programs may affect scale of operation as well. This issue has received relatively little attention.

Underwood and Caputo (1996) present a theoretical model of the impacts of deficiency payments, base acreage requirements, and set-asides on farmers' allocation of land between program and non-program crops and of pest control measures between pesticides and alternative knowledge-based pest control methods.

Leathers and Quiggin (1991) present a theoretical model of the impacts of price supports on risk-averse farmers' demand for inputs that may influence environmental quality. They show that a higher output price unambiguously increases demand for a risk-reducing input only if farmers exhibit constant absolute risk aversion, while the effect of a higher output price on demand for a risk-increasing input is indeterminate under any assumption about risk preferences. If one considers only the effects of price and income supports on average output prices, therefore, little can be said qualitatively about the potential effects of these programs on environmental quality.

Beginning in 1985, the United States attempted to reduce potential environmental damage from agriculture by decoupling deficiency payments from current yields. Per-acre deficiency payment rates were thus determined by the difference between the target and market prices and by the amount of land enrolled in the program and thus eligible for payments. Of these two, only acreage is under the farmers' control. Recent research on environmental impacts of commodity programs in the United States has therefore focused on land allocation, that is, on extensive-margin effects. Policy discussions on environmental effects of commodity programs in the United States leading up to the 1995 farm bill focused increasingly on base acreage requirements as impediments to the use of crop rotations for fertility and pest management (see for example National Research Council 1989).

There have been few such empirical studies to date, none of which has examined base acreage requirements. Following Lichtenberg (1989), Wu and Segerson (1995) use county-level data from Wisconsin to estimate a logit model of shares of county farmland allocated to alternative crops as functions of output and input prices, land quality characteristics, and farm program parameters such as the target price of corn and acreage set-asides for corn, oats, and wheat. Their treatment of environmental quality effects is crude. Land quality characteristics are also used to construct an index of leaching vulnerability that classifies soils as having high, moderate, or low leaching potential.

Corn and soybeans are classified as "high-polluting," wheat and oats as "low-polluting," and hay as non-polluting. The estimated model is then used to simulate the effects of changes in feed grain program parameters (the target price and set-aside for corn) on crop acreage allocations and thus potential leaching. Plantinga (1996) similarly estimates a logit model of the shares of land in four land capability class groups in southwestern Wisconsin allocated to either dairying or forest as a function of the timber:milk price ratio. Conversion from dairying to forest is assumed to decrease soil erosion by amounts taken from 1987 National Resources Inventory data, and Ribaudo's (1989) estimates are used to value the water quality benefits of decreased erosion. Stavins and Jaffe (1990) combine U.S. Forest Service data on county-level forest and cropland acreage, average per-acre crop returns derived from the Census of Agriculture, Natural Resource Inventory data on natural flood and drainage conditions, and weather data with parametric assumptions about land quality distributions to estimate conversion of forested wetlands in the southeastern United States. Their model indicates that increases in average crop prices increase deforestation and wetlands loss substantially. Van Kooten (1993) uses data from a survey of Saskatchewan farmers to estimate a cost function for converting wetlands and native pasture to cropland. This cost function is combined with budget data under alternative price scenarios in a dynamic decision framework to estimate the effects of grain price supports on wetlands conversion over an 80-year period. Kramer and Shabman (1993) use budget data to estimate the net returns to conversion of wetlands to cropland under several agricultural policy and tax reform scenarios.

5.2 Price Stabilization and Crop Insurance

Agricultural programs may also be aimed at reducing price or income risk. Most developed countries also have policies aimed at stabilizing the prices of agricultural commodities. For example, in recent years the United States has set crop loan rates sufficiently low that they influence market prices only in exceptional circumstances. As a result, crop loans reduce price risk by giving farmers the equivalent of free put options. As is well known, reductions in price risk tend to increase agricultural production (Sandmo 1971), exacerbating environmental quality problems in situations where environmental quality and agricultural output are substitutes, and enhancing environmental quality in situations where the two are complements. Many countries also use crop insurance to protect farmers against catastrophic risk, e.g., major crop failure (see Hazell, Pomareda, and ValdJs (1986) or Hueth and Furtan (1994) for descriptions of these programs; elsewhere in this Handbook, Chambers discusses issues in the design of agricultural insurance).

As Chambers and Quiggin (1996) point out, when the government is risk-neutral and farmers are risk-averse, optimal environmental policy has two tasks: (1) providing farmers with insurance against income variability and (2) correcting incentives to ensure socially optimal crop choices and input allocations. Crop insurance may affect environmental quality by influencing both crop choice and input use (see Moschini and Hennessy elsewhere in this Handbook for a review of theoretical analyses). The little empirical literature available has shown that input use effects can be substantial. Horowitz and Lichtenberg (1993) found that corn growers in the U.S. Corn Belt who purchase crop insurance use significantly more herbicides, insecticides, and nitrogen fertilizer per acre and spend more per acre on pesticides than those who do not purchase crop insurance. Smith and Goodwin (1996) find that Kansas wheat growers who purchased crop insurance spent significantly less on fertilizers and on all agricultural chemicals than those who did not purchase crop insurance. A simulation study by Babcock and Hennessy (1996) suggested that crop insurance reduces optimal nitrogen application rates on corn in Iowa, albeit relatively little. The impacts of crop insurance on crop choice and total land in cultivation have not been investigated, nor have there been studies of input use effects in a broader variety of crops and locations.

5.3 Input Subsidies

As noted in Section 3, input subsidies influence environmental quality in two ways. On the intensive margin, they induce farmers to increase the use of the subsidized inputs. On the extensive margin, they induce shifts in output composition toward crops that use the subsidized input more intensively.

In developing countries, subsidies for agricultural chemicals (fertilizers, pesticides) have been blamed for exacerbating environmental quality problems ranging from impairment of human health (see for example Levine 1991 and Boardman 1986) to poisonings of fish and other aquatic wildlife to degradation of water quality (see for example Way and Heong 1994). Subsidies of this sort have been used widely in order to promote the adoption of Green Revolution hybrid crop varieties. Recent empirical work suggests that the net social benefits of these subsidies may be negative. Antle and Pingali (1994), for instance, find that the value of lost work time due to sickness caused by pesticide exposure in Philippine rice production outweighs the value of damage avoided. Heong, Escalada, and Mai (1994) argue that early-season insecticide use on rice does not reduce crop damage and may actually result in increased damage by suppressing invertebrate predators that control later-season insect infestations (see also Bottrell and Weil 1995). Grepperud (1995) argues that programs that enhance farm productivity or the productivity of off-farm labor may reduce farmers' soil conservation effort. Overall, however, empirical work on these issues is sparse.

In some cases, however, input subsidies of these kinds may actually enhance environmental quality. L\pez and Nikklitschek (1991) show that in an economy with both a modern agricultural sector and traditional agriculture based on shifting cultivation, subsidization of inputs used in the modern sector can reduce deforestation by attracting labor from the traditional sector. In cases where it is infeasible to control entry into the traditional sector, fertilizer and/or pesticide subsidies may be an effective second-best policy for reducing adverse climate change effects, erosion and sedimentation of rivers, and species loss due to tropical deforestation.

In the United States, subsidies for irrigation water in the arid West have been a major source of environmental quality problems such as increased river salinity, heavy metal (selenium, arsenic) discharges into surface waters and consequent damage to wildlife, wildlife damage caused by reduced instream flows, and habitat loss due to conversion of land to agricultural uses. Water price distortions arise from a number of sources, including below-cost pricing of water from federal water projects, use of hydroelectric power revenues to offset the costs of providing irrigation water, exclusion of interest from capital repayment charges, and differential pricing for water delivered with lower reliability. Ultimately, however, these distortions continue to influence water demand primarily because of limitations on water marketing due to legal restrictions on water transfers. If water were freely marketable, farmers would face an opportunity cost of foregone revenue from potential water sales to higher-value users (urban areas, higher-value crops) regardless of the cost of acquiring irrigation water. This opportunity cost of water appears to be on the order of 2-5 times the current price that federal and state irrigation projects charge for irrigation water. Institution of water markets would thus likely increase farmers' opportunity cost of water substantially, leading to reductions in drainage and saline effluent through reductions in water application rates, shifts to less water-using crops, and investment in more efficient irrigation technologies.

When drainage and effluent are increasing functions of water application rates, as is typically the case, an increase in the price of water can function as the equivalent of a Pigouvian tax. Linear programming (Horner 1975; Gardner and Young 1988) and econometrically based simulation studies (Caswell, Lichtenberg, and Zilberman 1990; Dinar and Letey 1991; Weinberg, Kling, and Wilen 1993; Weinberg and Kling 1995) indicate that increases in water prices (and thus institution of water markets) could alleviate drainage and saline effluent problems substantially through shifts in cropping patterns, changes in irrigation technology, and reductions in water application rates for given crops and irrigation technologies. Moore and Dinar (1995), however, argue that water may be quantity-rationed at present in many areas, that is, that farmers presently are willing to purchase more water than is available at current prices. In such situations, increases in water prices can result in smaller than anticipated reductions in on-farm water use. They present econometric evidence indicating that California farmers using water from the Central Valley Project may be quantity-rationed.

Creation of water markets or other institutional reforms that price water at its true value may be effective means of controlling environmental damage in irrigated agriculture caused by other inputs as well. Larson, Helfand, and House (1996) note that increases in the price of water will reduce the use of complementary inputs and therefore environmental damage from the use of those inputs. They present simulation results indicating that taxing irrigation water achieves given reductions in nitrate leaching from lettuce at per-acre costs roughly equal to those required under combined water and nitrogen taxes.

Government investment in infrastructure and other public goods may implicitly provide subsidies for agricultural inputs that influence environmental quality. Stavins and Jaffe (1990) argue that federal drainage and flood control projects have played an important role in fostering conversion of forested wetlands to agricultural use in the Mississippi Delta region. By reducing flood risk over broad areas, these projects implicitly lower the cost of land that would otherwise require significant private investment to make cultivation attractive. Their empirical model suggests that close to one-third of conversion of forested wetlands in the Mississippi Delta between 1934 and 1984 could be attributed to projects of this nature.

6. Concluding Remarks

As we noted in the introduction, the centrality of agriculture to human existence and the dependence of agricultural productivity on natural conditions give environmental quality problems in agriculture certain distinctive features. Agriculture's dependence on natural conditions creates incentives for resource stewardship that may incidentally induce farmers to protect environmental quality. This dependence also means that heterogeneity and uncertainty are important factors in environmental policy design. The centrality of agriculture to human existence lies behind the presence of agriculture sector policies in virtually all countries. As a result, interactions between agricultural and resource policies play important roles in determining the effects of environmental policies in agriculture.

We began with a discussion of stewardship in order to answer the question of whether environmental regulation is needed in agriculture. If stewardship incentives are sufficiently strong, it is possible that farmers' pursuit of their own self-interest will result in adequate provision of environmental quality. Education and exhortation should suffice to fill any gaps by bringing unnoticed opportunities to farmers' attention. Such a view underlies the bulk of current environmental policy for agriculture in the U.S. The existing literature suggests that stewardship incentives are not sufficiently strong in most countries; the emergence of environmental problems association with agriculture is *prima facie* evidence that they are not. Therefore, there is a need for some form of environmental regulation. The literature also suggests that promotion of more environment-friendly farming methods does not always enhance environmental quality. New precision application technologies that reduce input requirements on the intensive margin may have extensive margin effects that impair environmental quality, so that introduction of a new precision application method may worsen environmental quality overall. Information-intensive technologies may similarly worsen environmental quality by increasing the use of polluting inputs. Even when these technologies do help improve environmental quality, the extent of improvement is likely to be less than anticipated. In general, careful empirical analysis is needed to determine the likely effects of these technologies. Relatively little has been conducted to date, however.

While more traditional forms of regulation (economic incentives, standards) appear to be needed, the empirical basis for setting such standards is generally lacking. The bulk of the empirical literature relies on farm-level simulation models. There is considerable evidence that these models predict ambient environmental quality poorly at scales of interest to policymakers, suggesting that alternative approaches are needed. Approaches involving interdisciplinary collection and analysis of data linking ambient environmental quality and agricultural production seem the most promising.

One interesting sidelight in this discussion was the claim of the sustainable farming advocacy literature that environmental regulation in agriculture might resuscitate traditional family farms in developed countries like the U.S. The formal argument is that once the full social value of environmental quality is taken into account, the joint costs of agricultural output and environmental quality are lower in smaller-scale integrated crop/livestock operations than in larger-scale specialized crop or livestock operations. While this proposition seems to enjoy considerable currency in policy discussions (at least in the U.S.), it has not been analyzed rigorously either theoretically or empirically.

We then turned to the question of the appropriate instruments for environmental regulation in agriculture. Developed countries mainly use a combination of direct regulation (imposition of best practice standards) and subsidies for the adoption of best practice farming methods to deal with both adverse and beneficial environmental effects. Emissions-based regulation tends to be infeasible given the high cost and technical difficulty of monitoring associated with nonpoint source problems like those typical in agriculture. Our analysis of regulation in a heterogeneous industry implied that negative (positive) incentives applied to inputs were preferable for cases involving adverse (beneficial) environmental effects because they gave farmers the proper signals on both the intensive and extensive margins. Our analysis of the choice of policy instruments under uncertainty was inconclusive due to lack of information about the relative sensitivity of crop productivity and environmental damage to uncertainty, although the fact that most of the inputs associated with environmental quality problems are risk-increasing argues for the superiority of incentives. Input taxes (subsidies) differentiated across farm types (e.g., crops, soils, slopes, location) seemed feasible to implement in many cases where inputs have adverse (beneficial) environmental effects. The major exception was fertilizer, which we discuss further below.

We also examined the design of environmental regulation under hidden information (adverse selection) and hidden action (moral hazard). Most studies of problems involving hidden information have neglected issues of contract enforcement such as compliance monitoring and secondary markets. The difficulty of monitoring compliance and the ease with which secondary markets can arise suggests that hidden information is relatively intractable for most environmental quality problems in agriculture. The major exception is land use. Hidden information is likely to be an issue in local-scale land use policies (farmland preservation) but can be avoided when the number of potential participants is sufficiently large (e.g., national land retirement programs). Very few studies have examined environmental policy design in cases involving hidden action. Most have examined cases where emissions were either observable directly or could be inferred from observation of ambient environmental quality and thus have little applicability to nonpoint source problems.

Emissions trading would seem an attractive way to implement incentive-based regulation of environmental quality in agriculture. However, there has been very little research on emissions trading under conditions characteristic of agriculture, namely when monitoring emissions is excessively costly, when the environmental quality effects of alternative farming practices vary across farms, and when there is significant uncertainty about those effects. Further research is clearly indicated given the growth of interest in such programs.

Of the inputs associated with environmental quality problems, fertilizers appear to be the least amenable to regulation. The environmental quality effects of fertilizers vary according to crop type and natural production conditions in ways which first-best taxes or best practice standards cannot take into account. Second-best taxes or best practice standards are unlikely to be enforceable due to the ease of creating secondary markets and the difficulty of monitoring compliance. The empirical literature suggests that nitrogen and phosphorus, the nutrients most commonly associated with pollution problems, are risk-increasing, so that price and income stabilization programs tend to increase their use. This latter finding is troubling because countries like the U.S. tend to keep stabilization and insurance programs in place even while phasing out price and income support programs. Moreover, to the best of current knowledge, nutrient pollution of surface and ground waters tends to be the most widespread environmental quality problem associated with agriculture. These considerations suggest that creative thinking about policy design is especially needed for dealing with nutrient problems.

The importance of understanding interactions between agricultural, natural resource, and environmental policies is well established by now. Careful theoretical analyses show that these interactions are frequently too complex to permit simple unambiguous generalizations, e.g., about the effects of price support or price stabilization programs on environmental quality. As a result, empirical studies of these interactions are especially important for policy design. Unfortunately, sound empirical studies are generally lacking.

One final set of issues generally neglected in the literature to date relates to environmental quality effects of marketing. Most studies of environmental quality problems in agriculture assume a simple, two-sector competitive agricultural economy consisting of price-taking farmers and consumers. In developed countries, at least, processors and marketers (wholesalers, retailers) account for a large share of the food sector. Much of the interaction between farmers and their immediate customers is conducted under some form of vertical coordination, ranging from contracts to integration of operations. In many cases, processors and growers supply some production inputs, so that agricultural output and environmental quality are produced jointly by both sets of agents. Vertical interactions can play an important role in environmental quality problems even in industries without explicit forms of coordination. For example, quality standards, which can arise from informational problems encountered in marketing, can influence inputs such as pesticides (Babcock, Lichtenberg, and Zilberman 1992; Starbird 1994; Lichtenberg 1997). To date, research on environmental quality problems in agriculture has concentrated on situations involving independent farms, ignoring the possibility that numerous agents jointly produce both environmental quality and agricultural output. Both conceptual and empirical research could help improve environmental policy design in such industries.

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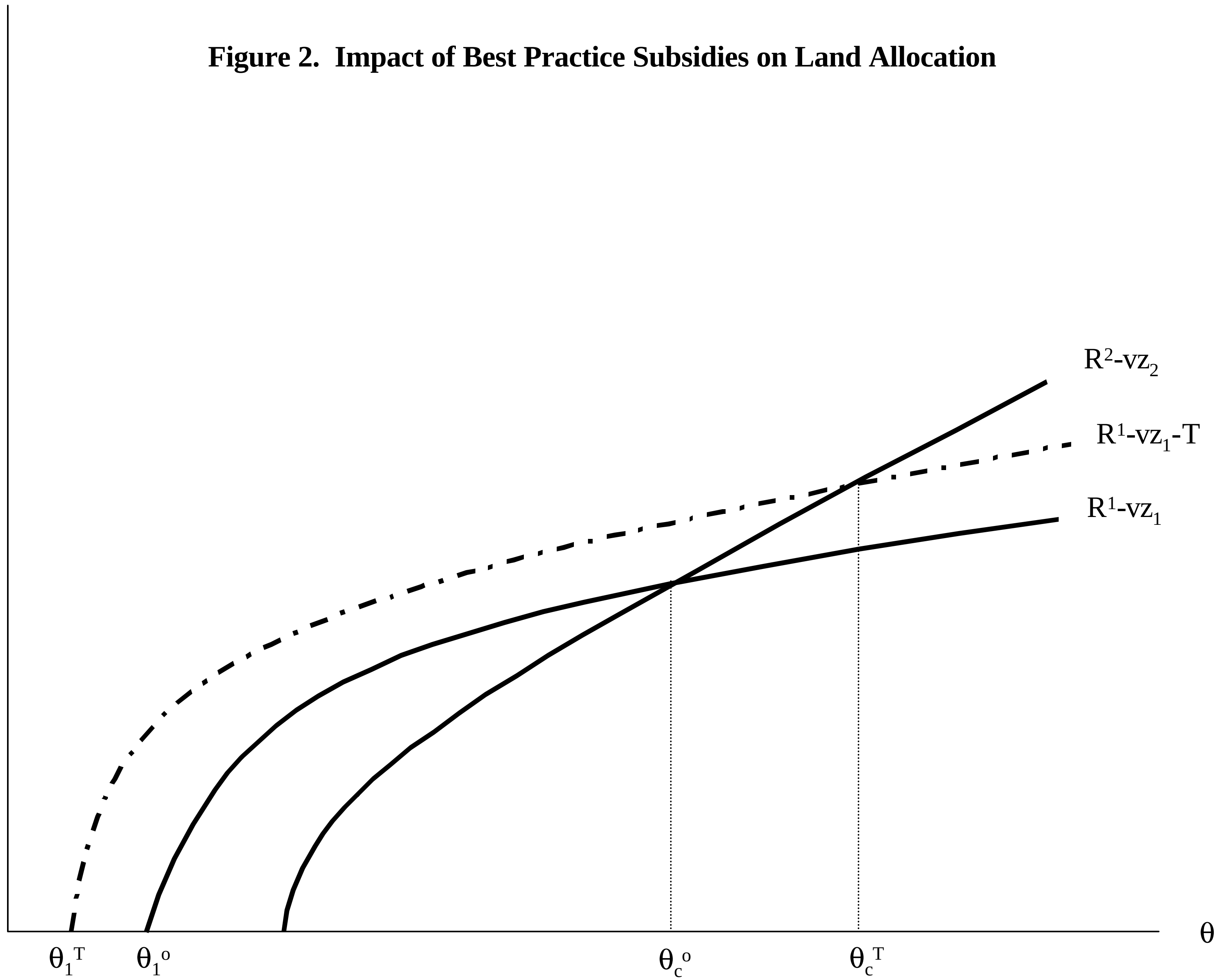
Figure 2. Impact of Best Practice Subsidies on Land Allocation



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