The Economics of Pesticides and Pest Control*

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ABSTRACT

Pesticides have been a major contributor to the growth of agricultural productivity and food supply. Yet, they are a source of concern because of human and environmental health side effects. This paper presents methodologies for assessing the productivity and health effects of pesticides. It also provides an overview of some of the major empirical findings. This paper covers major research that analyzes alternative approaches to address issues of resistance buildup, risk and environmental and human health, predator–prey relationships, as well as dynamic considerations. The paper summarizes existing policies that vary from the prescribed social optimum suggested by economic theory to those motivated by political–economy factors and risk aversion. Analysis is provided to relate pesticide policies to the larger context of agricultural and environmental management. This paper also presents recent modeling of invasive species and agricultural biotechnology.

1 INTRODUCTION

Growing concern about environmental protection, human health, and food safety has brought renewed interest in pesticide use in agriculture. This interest has largely

manifested itself in the form of increased government regulation of pesticides and has given rise to a market for organic foods. But with several regions of the world suffering food shortages and per capita food production in decline, and with the loss of agricultural land to other uses, there is a continued need for improving agricultural productivity. The food demand of an ever-growing world population, combined with interest in biofuels as an alternative to fossil fuels, will put increased pressure on agriculture and require further productivity gains. Pesticides were a key factor in significant agricultural productivity growth during the last century and continue to be a critical factor in reducing crop damage. Even as farmers explore alternatives to chemical pesticide applications, such as biological control and genetically modified (GM) crops, pesticide production remains a $32 billion industry with more than 5 billion pounds of pesticides applied to crops around the world every year (U.S. EPA 2007).

The world population is expected to grow 50 percent over the next 50 years to 9 billion people. This population growth, combined with the diet demands of a wealthier populace, are expected to double world food demand by 2050. Assuming production, regulation, and innovation trends of the past several decades continue, global pesticide production will be 2.7 times higher in 2050 than in 2000, exposing humans and the environment to considerably higher levels of pesticides. The demand for greater agricultural production poses a challenge perhaps as great as global warming. Agriculture constitutes the greatest driver of nonclimatic environmental change (Tilman *et al.* 2001).

The publication of Rachel Carson's *Silent Spring* in 1962 introduced the broader public to the risks of pesticides and fueled the ongoing movement away from pesticides and toward alternative damage control methods. Use of pesticides became controversial as consumers worried about risks to their own health from consuming products treated with chemical pesticides as well as environmental damage from pesticide applications. Those worries persist today, as do worries about the safety of farm workers. The ongoing debate over the use of genetically modified organisms (GMO) is also part of the movement begun more than 40 years ago.

For millennia, civilizations have relied on pesticides to protect their food supplies from damage. In the 21st Century, those pesticides have largely taken the form of chemical herbicides, fungicides, and insecticides that significantly improve agricultural productivity. Pesticide use has been in decline in several regions of the world, particularly developed countries (see Table 1). This trend is owed to increasing pesticide resistance, heightened concern about negative externalities, a better understanding of the costs and benefits of pesticide applications, and the development of new technologies and pest management strategies.

There exists today a portfolio of damage control tools at the farmer's disposal. These include biological control, which makes use of natural predators of pests and other natural phenomena like weather; mechanical control, which includes the use of specific tilling and cultivation techniques; and chemical control — the application of pesticides, herbicides, and fungicides. Toward the close of the last century, GMO emerged as another damage-abatement tool and a promising alternative to chemical pesticide applications. Genetic modification allows crops to be infused with pest-resistant genes, reducing the need for chemical applications.

Table 1. Pesticide consumption per hectare of agricultural land (Kg/Ha)

	1989–1991	1994–1996	1998–2000
Developed countries	0.64	0.54	0.52
Developing countries	0.18	0.25	0.28
World	0.40	0.38	0.39

Source: FAO Compendium of Agricultural-Environmental Indicators, 1989–2000.

Economics can be used to inform both private and public decision makers about trade-offs in the use of pesticides and other pest control strategies, enabling them to make decisions that help them reach their objectives. Much of the pesticide economics literature has focused on determining when pesticide use improves social welfare. Such determinations rely on balancing the benefits of a pesticide application (less crop damage) with the total costs, including those borne by the farmer (price of pesticide and cost of labor and machinery for application) and those imposed on society (such as risks to human health and diminished environmental quality from pollution). Such analysis must consider a host of issues, including externalities and uncertainty, which we will address in this chapter. Despite some modeling difficulties and the need to consider wide-ranging effects of pesticide applications, economists have developed valuable methodological tools to inform decision makers.

The past several decades have been characterized by efforts to reduce toxicity and improve effectiveness of pesticides through the use of new technologies and better information. Integrated Pest Management (IPM) and agricultural biotechnology are two notable tools in this effort. They and others will be considered here, as will government attempts to reduce the social costs of pesticide applications and to protect humans from undue risk. Historically, such measures have been blunt and unsophisticated, achieving second- or third-best outcomes and imposing significant costs on innovation. In providing prescriptions for a more effective pesticide policy, we also note that such policies must be considered in conjunction with the agricultural policies — price supports, output limits, quality standards, etc. — that determine firm and market behavior.

In the first section, we present pesticides as a way to improve agricultural productivity and discuss several theoretical models for how pesticides enter the production function as well as empirical results derived under those models. We next discuss how randomness and uncertainty in pest populations and damage functions change firm decisions and lead to overuse of pesticides as a risk-reducing input. We also consider the effects of uncertainty in determining environmental and health effects. We then consider the challenge of resistance buildup and mechanisms for avoiding its onset. Next we analyze the policy framework for pesticide use and urge greater emphasis on the use of market-based incentives for reducing pesticide applications. We conclude with discussions of biotechnology and invasive species control, two areas that have received considerable attention in recent years and continue to be subjects of inquiry by agricultural economists.

2 IMPROVING PRODUCTIVITY BY REDUCING DAMAGE

Agricultural pests include animals, insects, plants, fungi, and bacteria that lead to a loss of crops or reductions in crop yield relative to potential yield that would be possible in a world without pests. Pest damage can also reduce crop quality and may include blemishes on produce that reduce the value of agricultural commodities. Pest damage has afflicted growers since the beginning of agriculture, and the history of efforts to reduce pest damage is equally long. Only in the past century have growers benefited from chemicals that kill or deter pests. The use of chemicals has been the dominant paradigm for pest control in recent decades and has succeeded in greatly reducing crop losses.

Agricultural productivity has improved greatly in the past century, and particularly the past 50 years. Consider that while the world population more than doubled in the past 50 years, per capita food production increased; today, more than 6 billion people consume a daily average of 2,700 kcals per person compared to the daily average of 2,450 kcals consumed by each of the 2.5 million people in 1950 (WHO/FAO 2003). In each of the past four decades, per capita crop and livestock production increased by roughly 0.5 percent (FAO 2004). The significant gains in productivity can be largely attributed to improved crop varieties, new irrigation and harvesting technologies, and to developments in chemical fertilizers and pesticides.

Though pesticides are a major contributor to the ability of the agricultural community to feed a growing world population, they impose a variety of costs on society. Economic analysis of pesticide use distinguishes between social optimum and private optimum. The socially optimal level of pesticide use is an outcome of maximizing the net benefit to society that includes net benefit to consumers, farmers, producers of chemicals, as well as the environment. Different agents (farmers, chemical companies) make their pesticide choices to maximize their particular private net economic benefit. For example, risk–neutral, price-taking farmers are assumed to determine their optimal level of pesticide use by maximizing their profits. Their choices may deviate from the social optimum if they do not pay for the externality cost of the pesticide. The analyses in this section will consider the farmers' choices of pesticide use, and the social optimum will be considered later.

The concept of economic threshold recognizes the importance of information on pest density and potential crop loss in the decision to use a pesticide. The economic threshold is defined as the pest density (or amount of plant damage) at which the marginal benefit of control just equals the marginal cost of control. The concept implies tolerance of some pests in the field as well as tolerance of the associated pest damage, such as less aesthetically appealing fruits and vegetables. As we will discuss in a subsequent section, farmers may not consider the positive and negative effects their pesticide use has on other farmers and on society at large. A farmer, therefore, would determine an economic threshold as the pest density at which his cost of applying pesticides just equals the value of reduced damage. Similarly, a social planner may determine a threshold at which the total cost of pesticide application to society is equal to the total benefit to society. Assuming the farmer obtains all benefits to his pesticide use decisions (which he may not, as we discuss later), and further assuming costs of pesticide pollution incurred by

society, the planner's threshold will define a pest density greater than that of the farmer's threshold.

Stern *et al.* (1959) note that detectable crop yield reductions do not occur in many crop–pest situations until a minimum "threshold" level of pest density is observed. The importance of marginal pest control, marginal pest damage, and timing of the pest control was introduced by Headley (1972). Hall and Norgaard (1974) provide a valuable extension to Headley's work, also considering the optimal dosage of treatments. At its simplest, the threshold model determines when treatment should and should not be undertaken and, in particular, posits that treatment should only be initiated when the value of pest damage rises above the social cost of pesticide application.

Determining the value of pest damage and the social cost of pesticides is difficult because relevant information is costly or impossible to obtain and because there is no consensus among economists as to how pesticide benefits should be determined. In particular, theory offers little guidance for how damage functions should be specified. Empirical work suggests the functional form of the damage function has a large bearing on the determination of socially optimal pesticide use (Lichtenberg and Zilberman 1986a; Fox and Weersink 1995).

Measuring pesticide productivity has been a contentious issue for the past several decades. In more recent years, this controversy has centered on whether GM plants should be widely adopted to combat pests. Early pesticide productivity estimates modeled pesticides as a production input that had a direct effect on crop yield. Using a Cobb–Douglas specification, early studies supported the conclusion that the actual value of marginal product of pesticides was greater than price, suggesting underuse of pesticides (Headley 1968; Campbell 1976; Carlson 1977). Such findings were contrary to conventional wisdom in the agricultural community that farmers overused pesticides so that the value of marginal product was less than price.

Modeling the effect of pesticides on crop yield is not straightforward because there are several specification issues. First, are pesticides a production input just like labor, capital, and fertilizer, or does their effect on crop yield warrant a different modeling approach? A popular model assumes pesticides enter a damage function and reduce crop damage rather than increasing crop yield through a production function. Second, can inputs be treated as strictly yield-increasing inputs or strictly damage-reducing inputs, or do some inputs belong to both categories? And finally, if we adopt the damage control framework, how do we specify the functional form of the damage function? We will set about to answer each of these questions in turn.

Lichtenberg and Zilberman (1986a) were the first to propose a damage control model for pesticides that acknowledge pesticides as a damage abatement input that has an indirect effect on output rather than a yield-increasing input, as modeled in previous studies. Damage control inputs constitute a class of production inputs that do not increase potential output and may, in fact, decrease it. Their contribution to production is increasing the share of potential output farmers realize by reducing damage from human and natural causes. In addition to pesticides, sprinklers for frost protection, feedlot immunizations and antibiotics, and windbreaks are damage control agents. They are quite distinct from other factors of production like land, capital, and labor. Apart from the

intuitive appeal of modeling pesticides as an input that reduces pest damage (rather than an input that increases potential output), the damage control framework has important economic ramifications.

The first implication accounts for the early econometric results that indicate under-use of pesticides at times when anecdotal evidence suggests overuse. The production function specifications used most often in factor productivity estimates imposed on the marginal factor productivity curve a shape that misspecifies marginal factor productivity for damage control agents, which decreases more rapidly over the economic range. This is illustrated in Figure 1, where w denotes constant marginal cost. As is easily seen, a specification that restricts the rate of decline of marginal factor productivity will overestimate the optimal level of pesticides.

The second contribution of the damage control specification is that it accounts for changes in damage control agent productivity over time. This is especially important in the case of pest damage abatement because of the adaptation of pests to control measures over time. Failure to model such inputs in the damage control context leads to incorrect predictions of behavior. Specifically, diminishing factor effectiveness reduces marginal factor productivity, which results in less input use. With pesticides, however, the opposite behavior is observed with the onset of pesticide resistance. Observed behavior is predicted by the damage control model: use of the damage control agent increases with resistance until resistance is sufficiently pervasive as to render alternative damage control measures more cost effective.
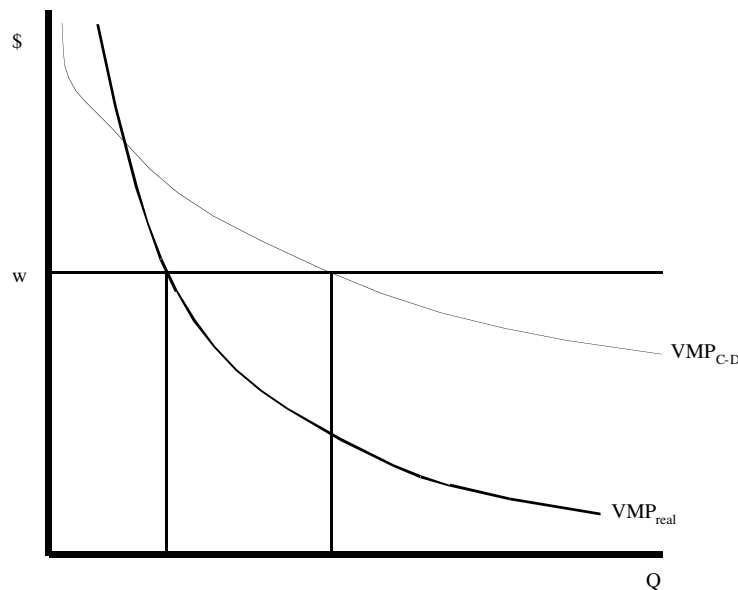

Figure 1. Upward bias in estimation of optimal pesticide use under standard Cobb–Douglas formulation

The damage control framework models output, Y, as a function of potential output, $g(Z)$, less damage, $D(N_1)$, where damage is a function of the size of the pest population and is measured as the percent of output lost to pest damage. Thus, output is given by

$$Y = g(Z)[1 - D(N_1)]. \tag{1}$$

In (1), Z's are inputs unrelated to damage control, such as labor, capital, fertilizer, etc. The pest population, N_1, is a function of initial pest population level, N_0, quantity of pesticide applied, X, and alternative damage control efforts (such as mechanical and biological control), A. Thus,

$$N_1 = h(N_0, X, A). \tag{2}$$

The function h in Eq. (2) is defined as the kill function and represents the size of the pest population after damage control measures have been undertaken. Clearly, h is decreasing in X and A.

A profit-maximizing firm will apply pesticides whenever the marginal cost of doing so at the interior optimum level of application is less than the marginal cost of crop damage. The socially optimal level of pesticide use, however, must account for social costs, in addition to the private costs facing the firm. Thus, if total damage from pests at the interior optimum is less than the social costs associated with an application of pesticides, then the welfare-maximizing level of pesticides is zero.

If we let p denote output price, u the price of inputs unrelated to damage control (Z), w the price of a unit of pesticides (X), and v the price of alternative control methods (A), then the optimization function of a profit-maximizing farmer is:

$$\underset{\{Z, X, A\}}{\text{Max}} \, pg(Z)[1 - D(h(N_0, X, A))] - uZ - vA - wX, \tag{3}$$

where the distinction between pesticides and alternative control methods is made to emphasize the trade-off between pesticides and other control options, such as biological control.

The model predicts that pesticide use will rise whenever output price, potential output, and the price of alternative controls increase. It will also rise with a fall in the price of pesticides.

The optimal pesticide application is where the value of the marginal benefit of damage control $-pg(Z)\frac{\partial D}{\partial N_1^*}\frac{\partial h}{\partial X}$ is equal to the price of pesticide application w. N_1^* is the optimal population of pests after application.

The Lichtenberg and Zilberman model is popular because it explicitly incorporates a damage function and because it has been successfully applied. Yet, critics question restrictions imposed on the effects of pesticide inputs. Oude Lansink and Carpentier (2001) challenge the partition of inputs into those that reduce damage and those that increase productivity (see also Carpentier and Weaver 1997; Oude Lansink and Silva 2004). They present evidence of interactions between the two categories. Oude Lansink and Silva (2004) also critique implicit assumptions imposed on parameters in the damage control model as well as the assumption of a nondecreasing damage control function. While the critiques are not without merit, and whereas the damage control model may

not be applicable in every case, its general validity and applicability make it a considerable innovation. Further, data limitations render modeling of the input interactions proposed by Oude Lansink and Silva intractable.

The introduction of a damage function poses another problem for researchers. Economics offers little insight into what functional form should be used to specify the damage function. Uncertainty surrounding the appropriate specification of the damage function suggests a role for interdisciplinary work between economists and agronomists to better model the damage function. Alternatively, a variety of functional forms can be used to bound productivity estimates. In addition, functional forms will vary by scale, so that different functional forms may be appropriate when modeling damage to a field versus a farm or region.

Not withstanding the difficulty in specifying functional forms, the damage control model has been applied by Carrasco-Tauber and Moffitt (1992), Chambers and Lichtenberg (1994), and Oude Lansink and Carpentier (2001), producing varying results, some of which indicate overuse of pesticides. Varying assumptions by the authors about parametric functional forms may account for some of the variance in productivity estimates. Carrasco-Tauber and Moffitt, for instance, consider logistic, Weibull, and exponential forms and obtain marginal product estimates for pesticides of $7.53, $6.88, and $0.11, respectively. Because economic theory behind the specification of the damage control function is weak, studies often consider a variety of functional forms. Exponential forms are common in pesticide kill functions, but there is no theoretical basis for preferring any one form to another (Praneetvatakul *et al.* 2002).

Praneetvatakul *et al.* (2002) employed a damage control model with logistic, Weibull, and exponential forms, to estimate marginal productivity of pesticides in Thailand rice production. They compared those results with estimates from the conventional Cobb–Douglas model that incorporates pesticides in the production function. They confirmed that the conventional (Cobb–Douglas) approach overstates the marginal product of pesticides. They determined pesticides are overused in rice production. Each of the three specifications of the damage control function is found to produce a satisfactory fit and a positive result for the effect of pesticides on output. The authors found, however, that the exponential damage-control function produced results similar to the Cobb–Douglas estimates, which they and others have concluded overstate the effectiveness of pesticides. They confirm the Carrasco-Tauber and Moffitt (1992) result that the specification of the damage control function can significantly impact results.

The damage control model has been especially useful in shedding light on the economics of GMO, specifically the ability of GMO to reduce pesticide applications and increase yield. GM crops are a hotly contested modern input that makes some policymakers uneasy, principally because of uncertainty surrounding the impact their widespread adoption would have on ecology and food safety. Not withstanding these concerns, GM crops have been demonstrated to greatly increase yields and decrease pesticide applications in developing countries even though their impact in developed countries is more muted. In the United States, for instance, a relatively small 10 percent yield increase is attributed to insect-resistant cotton, and the yield benefits of insect-resistant corn are negligible (Carpenter *et al.* 2002; Pray *et al.* 2002). However, in India, farms that planted

cotton containing the gene to produce naturally occurring pesticide *Bacillus thuringiensis* (Bt) reduced their pesticide applications by 70 percent and benefited from yields 80 percent higher than those of traditional crops (Qaim and Zilberman 2003).

Huang *et al.* (2002) found similar results in China and used the damage control model to estimate the marginal product of cotton containing the Bt gene (Bt cotton). They estimated a return of $5.96 per $1.00 expenditure on Bt cotton. They also found Bt cotton reduced pesticide use by 70 percent. In Argentina, the damage control model suggests that Bt technology has an average yield effect of 32 percent, based on farm level surveys completed in 2001 (Qaim and de Janvry 2005). The significant productivity benefits of Bt crops reflect low pesticide use and high pest pressure in Argentina. Their results confirm the proposition of Qaim and Zilberman that the gains from GM crops will be most significant in regions where pesticides are not already effectively controlling pest damage (developing countries) and where pest pressure is high (tropical and subtropical regions). Similar effects of Bt cotton in Mexico (Traxler *et al.* 2001) and South Africa (Thirtle 2003) serve to reinforce the potential for widespread adoption of GM crops in developing regions. A further discussion of agricultural biotechnology, which we use synonymously with GM, follows in the final section of this chapter.

2.1 Quality Effect

It is important to note that while the preceding discussion emphasizes the yield–increasing benefits of pesticides, chemical pesticides can also have a quality-improving effect. This is particularly true for fresh produce, where pest damage can render some output unmarketable or require it to be sold at lower quality grades and accordingly lower prices. Where pesticides have quality effects, those effects should be included in the determination of profit-maximizing pesticide use and socially optimal levels of pesticide use. Produce that qualifies for sale on the fresh market often sells at a premium relative to that sold in the processing market, giving producers' strong incentive to improve quality. For example, fungicides applied to the North Carolina apple crop are demonstrated to improve both yield and quality while insecticides are observed to have only a quality-improving effect (Babcock *et al.* 1992). In fact, all insecticide use and 15 percent of fungicide use by apple producers in North Carolina are driven by quality considerations. Babcock *et al.* find that even when quality is considered in the analysis, chemical agents are overapplied to the apple crop relative to the profit-maximizing level.

2.2 Conclusion

The determination of the net benefits of pesticide use is central to optimal use decisions and, therefore, the regulation of pesticides. The damage control framework provides an appealing way to model the net benefits of pesticides that is supported in empirical work. Because there is yet little consensus as to the proper form of the damage control function, it is clear economists must work with agronomists and biologists to arrive at appropriate functional forms that recognize the underlying biological processes that determine pest control and crop damage. And, as stated earlier, the economic threshold suggests some

level of pests and crop damage should be tolerated so that applications only occur when the benefit of an application equals the total social cost of an application.

3   ECONOMIC RISK AND PESTICIDE USE

Pest arrival is an uncertain event. Crop damage is subject to stochastic processes. And pesticide productivity varies across time and space. There is, therefore, considerable uncertainty about farmer profits with the use of pesticides. This uncertainty, as we will see, can lead to overuse of pesticides relative to the social optimum. Following the model defined above by Eqs. (1)–(3), we can illustrate the varieties of uncertainty involved.

Pest density, N_1, is a function of a kill function $h(\cdot)$ as given by (2). Recall it depends on the initial pest density (N_0) and on the level of chemicals applied (X) and other alternative control methods (A). The kill function varies for different pests and pesticides and depends on timing and environmental conditions. Total output (Y), given by (1), is a function of pest-free yield, $g(Z)$, where Z denotes production factors, and $D(N_1)$, a damage function that depends on a range of factors and varies for different pests and different crops or pastures. Profit, given by Eq. (3), is determined by output price, p, unit pesticide cost, w, as well as other production costs not explicitly modeled.

The model includes a number of elements that are likely to be stochastic. In Eq. (2), the pest density, N_1, may be uncertain because the initial pest density, the proportion of pests killed, or the chemical dosage actually applied are unknown or are estimated with error. In Eq. (1), the yield, Y, will be uncertain due to uncertainty about pest-free yield, the level of damage, and the final pest density. The profit in Eq. (3) is likely to be affected by variance in yield and output price.

In addition, there are other random factors not explicitly contained in the model. The extent of pest infestation, weather, prices, biological factors, damage per pest, pesticide effectiveness, and other random variables affect pesticide productivity and profits. There is widespread consensus in the literature that, in many circumstances, risk considerations influence pesticide use (Carlson and Main 1976; Conway 1977; Reichelderfer and Bottrell 1985; Tisdell 1986). Many authors view pesticides as a form of insurance for farmers. Feder (1979) provides a model to determine the effects of uncertainty on the pesticide-use decisions of risk-averse farmers. His theoretical approach suggests that an increase in the extent of uncertainty about pest damage increases the volume of pesticide applications for any given pest population and any given marginal control cost. This is true despite the fact that average pest damage is unchanged with increased uncertainty. Reichelderfer (1980) and Wetzstein (1981) go so far as to claim that risk reduction is the main motivation for pesticide use.

The risk effects of pesticides are important in a number of contexts. For example, crop insurance has been proposed as a means of reducing pesticide use, based on a belief that pesticides are risk reducing (Miranowski 1974; Carlson 1979). It has also been suggested that farmer response to policies such as pesticide taxes will depend on the risk effects of pesticides (Leathers and Quiggin 1991).

Numerous theoretical and empirical papers have discussed these issues directly and indirectly (see Pannell 1991 for a nice survey on this topic). Horowitz and Lichtenberg (1994) provide a general model to evaluate pesticides as risk-reducing or risk-increasing inputs. They consider a production technology given by $f(X, \varepsilon)$ where X is an input, ε is a random variable, and $f(X, \varepsilon)$ is a crop output. The parameter ε is considered to be an index of random factors affecting production that are independent of X. Production is increasing in ε. Therefore, in general, $f_X(X, \varepsilon) > 0$ and $f_\varepsilon(X, \varepsilon) > 0$. If we let p be the nonstochastic price per unit of output and w the unit cost of input X, then state-contingent farm profits are $pf(X, \varepsilon) - wX$. We assume a farmer will chose X to maximize his expected utility: $\mathrm{Max}_{\{X\}} E[U(pf(X, \varepsilon) - wX)]$, where utility is denoted by U and is a function of profits.

Whether an input X is risk reducing or risk increasing depends on $f_{X,\varepsilon}(X, \varepsilon)$. An input X is risk reducing if $f_{X,\varepsilon}(X, \varepsilon) < 0$, since the input raises production more in bad states of nature than in good states. That is, the marginal product of X is less in good states than bad states of nature, which has the effect of reducing the variance of output between good and bad states. The input is risk increasing if $f_{X,\varepsilon}(X, \varepsilon) > 0$. Quiggin (1991) characterizes an input as risk reducing if a risk-averse producer would use more of it than a risk-neutral one or if a producer with output insurance would use less of the input.

Horowitz and Lichtenberg (1994) consider three scenarios of uncertainty: (1) uncertainty about pest damage only; (2) uncertainty about crop growth conditions only; and (3) uncertainty about both damage and crop growth conditions. They show that when the principal source of uncertainty is pest population, pesticides are likely to be risk reducing, confirming the conventional view. However, if crop growth is also random and if pest populations are high primarily when crop growth conditions are good, then pesticides will likely be risk increasing, because pesticides increase output in already good states of nature. They thereby increase the variability of harvests.

The damage function may differ across stages of crop growth and may not remain constant over pest densities. Also, potential yield and future pest densities for the season may not be known at the time the pesticide-use decision must be made. The mean levels of these parameters may be used to calculate the threshold level (defined in the first section of this paper), but the resulting estimates may be imprecise. Nevertheless, extensions of the economic threshold model that include sensitivity analysis for changes in economic and biological parameters may provide useful starting points for recommendations regarding the amounts of pesticides to be used (Marra and Carlson 1983; Talpaz and Frisbie 1975).

A recent survey by Davis and Tisdell (2002) outlines more complicated economic threshold models to aid those making pest management decisions. Some limitations for applying economic threshold models are raised, such as the effect of farmer risk aversion and the knowledge required of farmers about cost and damage functions. In addition, the authors discuss complexities in the nature of yield loss functions due to uncertainty about pest densities, the effects of multiple pests, and the occurrence of pesticide resistance. We will consider pesticide resistance later in this chapter.

3.1 Preventive Applications

Conventional wisdom, particularly in the medical profession, is that "an ounce of prevention is worth more than a pound of cure." In the world of pesticides, however, the stochastic nature of pest populations often renders responsive applications of pesticides superior from a social standpoint to preventive applications. Responsive applications are defined as those that take account of the actual pest population obtained in each crop cycle, whereas preventive applications occur before pest populations are realized and are based on expectations of population size.

With preventive application, pesticides are applied without an attempt to determine potential pest populations. Instead, based on experience or historical data, the farmer makes educated guesses about the likelihood of various pest population levels occurring. The farmer then chooses a level of pesticide use to maximize expected profit. For example, suppose pest population, n, is a continuous random variable with known distribution. In this case, the farmer's problem is to maximize expected profits from preventive application, $\prod_p$, with respect to pesticide use (X), which does not vary in n

$$\underset{\{X\}}{\text{Max}} \prod_p = pg(Z) \int_{N_1}^{N_2} \left[1 - D(\overline{N}, X)\right] \psi(\overline{N})dN - wX,$$

where $\psi(N)$ is a density function of pest population defined on a biologically determined support N_1, N_2, where the lower bound of the support can be considered the minimum number of pests needed to ensure reproduction (which requires that potential mates locate each other), and the upper bound can be thought of as the carrying capacity for the pest. $\overline{N}$ is used to denote expected pest population level.

Although the pest population is uncertain, the level of pesticide application, X, will be the same with preventive application, regardless of which pest population level, N, actually occurs. This result follows directly from the definition of preventive application — pesticide-use decisions may be based on lagged indicators and a "guess" about the population size, but not on current information, particularly not the current realization of pest population size.

3.2 Responsive Applications and Integrated Pest Management

Responsive applications are the alternative to preventive applications and require farmers to monitor pest populations and apply chemicals according to observed pest levels. Responsive applications are a key component of modern IPM programs. IPM includes an assortment of techniques at the disposal of the producer that is designed to maintain pest infestation at an economically acceptable level rather than to eradicate all pests. It emphasizes the use of technology to permit decision making with increased information. It also integrates biological, cultural, and chemical control methods. IPM requires: (i) planning for the prevention or reduction of pest problems; (ii) learning pest identification; (iii) monitoring of crops, pests, and local environments; (iv) determining damage thresholds that delimit acceptable levels of crop quality and yield; (v) coordinating genetic, cultural,

physical, biological, behavioral, and chemical control methods to optimize and synergize their effects; and (vi) evaluating and adjusting IPM to strive for improved control.

With responsive applications, farmers incur a monitoring cost, which we will denote as m. The farmer's problem is to maximize profit from responsive application, $\prod_r$, with respect to $X(N)$, which varies according to observed pest level

$$\underset{\{X(n)\}}{\text{Max}} \prod_r = pg(Z) \int_{N_1}^{N_2} [1 - D(N, X(N))] \psi(N)\, dN - w \int_{N_1}^{N_2} X(N)\, \psi(N)\, dN - m.$$

The farmer will choose responsive or preventive applications depending on which offers higher expected profits. He faces a trade-off between monitoring costs and pesticide cost savings made possible with monitoring. Formally, he will choose the greater of $\prod_p$ and $\prod_r$. Given costless monitoring, preventive applications will always be suboptimal. However, if the costs of monitoring are high, it may be socially optimal to forgo monitoring and undertake preventive pesticide applications.

If the difference between optimal pesticide applications under different pest levels is large, and monitoring cost, m, is relatively small, then responsive application will give a higher level of expected profits than preventive application. However, even if the variance of $X(N)$ is large, farmers may still use preventative application if the price of pesticides is relatively cheap or the cost of IPM is relatively high. Technology can be used to reduce the costs of monitoring and thereby induce more responsive applications.

IPM is not one technology, but rather encompasses many practices and components. Wiebers (1992) shows that there are numerous different types of IPM practices in use in California and each addresses different pests and different stages of production. For example, a farmer may use predators to deal with insects during certain stages of production, but not others. Some farmers use an economic threshold approach for one pesticide and adopt a preventive approach for another (Wiebers 1992).

IPM is information intensive because both pests and beneficial organisms need to be monitored (Hall and Duncan 1984). Scouting is the primary method of tracking pest populations and involves regular and systematic sampling of fields to estimate pest infestation levels. Field scouting costs range from $3.00 to $5.00 per acre for field crops to as much as $20 per acre for high-value crops for which pest management is particularly important to profits. The second component of IPM is the use of economic thresholds, where treatment decisions are based on economically derived decision rules.

Although IPM is sometimes defined as an attempt to reduce pesticide use while maintaining current production levels, the empirical evidence on the effect of IPM on pesticide use is mixed, even for a given crop. Some econometric studies find that IPM adoption leads to a significant reduction in pesticide use (Burrows 1983; Fernandez-Cornejo 1996, 1998), while others find an increase in use (Yee and Ferguson 1996), and still some others find no significant effect (Wetzstein *et al.* 1985; Fernandez-Cornejo and Jans 1996). Empirical results on the impact of IPM on pesticide use may not be uniform because of differences in the operational definition of IPM, particularly the practices considered in the IPM "bundle" (Fernandez-Cornejo and Ferraioli 1999). For example, scouting of cotton fields is found to increase pesticide use when scouting effects

are studied in isolation. But when considered as part of a larger IPM package, the IPM package is found to reduce pesticide use (Norton and Mullen 1994; Yee and Ferguson 1996). Taylor (1980) showed that IPM adoption might lead to an increase in pesticide use if acreage increases as a result of adoption.

The evidence on the effect of IPM on yields or farm profits is also mixed, but it appears to be more uniform than the effect of IPM on pesticide use. Greence and Cuperus (1991) and Norton and Mullen (1994) provide a summary of empirical results of the effects of IPM for vegetables and for crops in general. At the farm level, IPM appears to reduce costs and increase net returns (Lacewell and Masud 1989; Ferguson and Yee 1993; Smith *et al.* 1987; Wetzstein *et al.* 1985). Several books provide a comprehensive review and case studies (Lutz 1998; Wiebers 1993; Kiss and Meerman 1991). Resosudarmo (2001) examines the impact of the Indonesian government's 1991–1999 IPM program and finds that it helped farmers reduce the use of pesticides by approximately 56 percent and increased yields by roughly 10 percent. Externality problems make it difficult for farmers to adopt IPM individually, so there exists a role for institutional support for collective action.

The extent to which IPM practices have been adopted varies by crop and region. This variability can be attributed to uneven distribution of technology, disparate public support, crop value variation, and different environmental conditions. IPM practices have been adopted around the world, particularly in developing countries, where excessive use of pesticides during the 1970s and 1980s caused serious health and environmental problems. IPM has had some successes in the industrialized world, but seen much more attention in the developing world, where the technology has been adopted with the help of extension efforts by OECD countries (Pretty 2005). Also, perceived risk may increase because of a perceived lack of knowledge and may be another factor deterring adoption of IPM. Both theoretical models and empirical research show that risk-averse producers optimally use less of a risk–inducing input than they would under certainty (e.g., Arrow 1971; Pratt 1964; Sandmo 1971; Anderson *et al.* 1977; Robison and Barry 1987). However, several papers (Greene *et al.* 1985; Hurd 1994) find that yield variability is not significantly affected by IPM practices.

Cowan and Gunby (1996) explain the continued dominance of chemical pest control by positive feedbacks associated with chemical control. In a competition among technologies, the presence of increasing returns to adoption can force all but one technology out of the market and the victor need not be the superior technology. In this context, path dependency of technology adoption can also explain limited adoption of IPM. Wiebers (1993) also attributes the continued dominance of conventional pest control to convenience and the limited marketing of IPM — a result of the technology not being embodied in a product. The U.S. Department of Agriculture suggests slow adoption is also the result of a belief among farmers that IPM is too complicated and difficult to use and that the recommendations of IPM often conflict with farmers' intuition.

Public support for IPM is derived, in part, from concerns about food safety and the environment, yet few studies have assessed the economic value of health and environmental benefits of IPM. Mullen *et al.* (1997) provide an approach for such assessment and apply it to the peanut IPM program in Virginia. They first estimate the effects of

IPM on environment and health risks posed by pesticides and then determine society's willingness to pay to reduce those risks. They estimate the total savings in external costs from the peanut IPM program at $844,000 for 59,000 households.

3.3 Endogenous Risk

The standard view of pesticide use generally presumes exogenous risk — that the likelihood of pest infestation is beyond the control of a producer, and pesticides act as market insurance against uncertain crop damage. The endogenous risk model suggests that a producer can invest in two types of risk-reducing technologies: self-protection and self-insurance. Self-protection reduces the probability of an undesired state of the world, while self-insurance reduces the severity if the state occurs (Ehrlich and Becker 1972). Archer and Shogren (1996) apply the endogenous risk model to the weed control problem. They model the herbicide application rate as investments in self-insurance which decreases the magnitude of yield loss, and model herbicide timing flexibility or herbicide persistence as investments in self-protection, which reduces the probability of a loss. This approach allows them to model both the type and quantity of herbicide used and to analyze the substitution between rate of application and persistence or timing flexibility. They find, for instance, that policies aimed at reducing herbicide loading can increase the use of more persistent herbicides. Similar endogenous risk frameworks have been widely applied to problems of invasive species, which we shall discuss in detail in a later section.

Models of endogenous risk incorporate evolution as part of the dynamic response to pest control strategy. Ewald (1995) suggests that the virulence of a disease is dependent upon the environment in which it lives. In particular, the virulence of diseases can be greater in areas where sewer systems and water treatment facilities do not exist. Whereas highly virulent strains of diseases may not persist because of their lethality and the speed at which they afflict victims, they may spread and survive where sewage flows into water systems that are used to supply drinking water. In such circumstances, the introduction of water treatment and sewage systems that prevent the spread of disease may lead to the evolution of strains that are less virulent. This example, although related to health, demonstrates an evolutionary approach that may explain the dynamics of resistance and a better perspective on predator–prey relationships. Further research on the interaction of evolutionary and economic dynamics may lead to improved assessment of pest control policies and help address invasive species control and similar problems (Ewald 1995).

3.4 Conclusion

As has already been mentioned, an individual farmer's pesticide-use decision will often not reflect a variety of costs imposed on society. Chief among these are environmental and human health costs and the buildup of resistance that can render pesticides useless after repeated applications. We have considered in this section the extent to which pesticide use is a response to uncertainty and risk and how reducing risk and uncertainty can lead to a reduction in pesticide applications. Additional uncertainty about external costs of

pesticide use and other information constraints make difficult the farmer's determination of his profit-maximizing pesticide strategies. Uncertainty is particularly prevalent in determining the health and environmental impacts of pesticide use, so we turn to that issue next, saving a discussion of resistance for the subsequent section.

4   HEALTH AND ENVIRONMENTAL EFFECTS

Impacts on environmental and human health increase the social cost of pesticides beyond the private cost facing the farmer. Pesticides are designed to kill organisms that share many biochemical pathways and physiological processes with nontarget species in the agro-ecosystem, including animals and humans. The biological commonalities make it difficult to develop pesticides that have ample margins of safety between the pest species and nontarget organisms. Furthermore, the removal of "pest species" can change the treated ecosystem in a variety of ways, including making it more susceptible to other pests (Knight and Norton 1989).

Careful analysis of the costs of pesticide applications requires not just a determination of impacts on the targeted pest species and crops, but also knowledge of species interactions, including effects on humans. Of primary concern to policymakers is agricultural worker exposure; consumer exposure through pesticide residues on foods; and damage to ecosystems from drift, runoff, and species interactions.

4.1   Pesticide Exposure

There is a growing literature on the control of the side effects of pesticide use. Parallel to a production function that relates the impacts of inputs and outputs, one can consider that the use of chemicals contributes to a risk generation function, where risk is defined as the probability of injury, mortality, or damage to wildlife. It is a combined outcome of four processes: application, contamination, exposure, and dose response (Lichtenberg and Zilberman 1988; Spear 1991). Based on risk assessment models, Lichtenberg *et al.* (1989) introduce models of the risk-generation function as the outcome of several processes that include both inputs (some of them policy parameters) and random effects.

Consider the following simple model: Let R denote the health risk of a representative individual. Taking application as given, R is the product of three functions:

 (i) $f_1(X_1, \beta_1, \varepsilon_1)$ is the contamination function and relates contamination of an environmental medium to activities of an economic agent (e.g., relates pesticide residues on apples to pesticides applied by the grower). β_1 is a parameter reflecting damage control activity at the site, which may include the use of protective clothing or other rules that preclude workers from entering fields for given periods of time following pesticide applications. X is a measure of pollution on site (e.g., the level of pesticide used), and ε_1 is a random variable.

(ii) $f_2(X_2, \beta_2, \varepsilon_2)$ is the human exposure function, which depends on β_2, an individual's actions to control exposure and a random variable, ε_2. For example, it may relate ingested pesticide residues to the level of rinsing and the degree of food processing

an individual undertakes. The product $f_2(\,\cdot\,)f_1(\,\cdot\,)$ is equal to the overall exposure level of an individual to a toxic material.

(iii) $f_3(X_3, \beta_3, \varepsilon_3)$ is the dose–response function, which relates health risk to the level of exposure of a given substance. It relates, for instance, the proclivity of contracting cancer to the ingestion of particular levels of a certain pesticide. It is based on available medical treatment methods, β_3, and a random variable, ε_3. Dose–response functions are estimated in epidemiological and toxicological studies of human biology.

Assume these factors interact multiplicatively, $R = f_1(X_1, \beta_1, \varepsilon_1)f_2(X_2, \beta_2, \varepsilon_2)f_3(X_3, \beta_3, \varepsilon_3)$. Assume further that the random elements in each function, ε_i, have different log normal distributions and are uncorrelated. Then, $\log R$ is distributed normally with mean equal to $\sum_i \mu_i(\beta_i, X_i)$ and variance equal to $\sum_i \sigma_i^2(\beta_i, X_i)$.

If we can estimate the functions $f_1(X_1, \beta_1, \varepsilon_1)$, $f_2(X_2, \beta_2, \varepsilon_2)$, and $f_3(X_3, \beta_3, \varepsilon_3)$, then we can determine the optimal combination of pollution control, exposure avoidance, and medical treatment using simple maximization techniques (Crouch and Wilson 1981; Lichtenberg and Zilberman 1988).

With this model, we can assess the risk from several exposure pathways. Chronic health risks from the use of contaminated drinking water can be modeled as the product of the level of contamination introduced into the environment, the rate at which the contaminant enters the water supply, rates of water consumption, and dose–response rates. Air-pollution-induced illness is modeled as a product of emission rates, air intake rates, and dose–response rates. And pesticide residue poisoning is the product of pesticide application levels, oxidation rates, dermal absorption rates, and dose–response rates.

As an illustration, suppose a chemical that reaches groundwater may cause human disease. The number of average sick days lost due to use of the chemical, for example, reflects the volume of product applied, the percentage of the product that percolates to groundwater (contamination), the number and types of people who drink the water (exposure), and the health impacts of various exposure levels (dose/response). The first three of the four processes are intuitive. The dose/response process translates exposure to the probability of contracting certain diseases and includes acute (immediate) and chronic (due to repeated exposure or risks that develop over time) risks of poisoning.

The side effects of pesticide use vary significantly over time and space. For instance, the environmental damage associated with chemical applications in a riparian zone is often much greater than applications in other areas because sensitive habitat and water resources may be poisoned by pesticide drift and runoff. Similarly, impacts of the same level of application may be significantly different during a dry period versus a period of precipitation (Lichtenberg *et al.* 1989). This heterogeneity accounts for some of the uncertainty that surrounds health and environmental risks. Because these effects tend to be considerably uncertain, evaluations of environmental and health regulations, using benefit-cost analysis and employing estimates based on average risk, tend to be misleading. They completely ignore uncertainty. Such policy analyses need to deal not just with risk but also with risk compounded by uncertainty.

4.2   The Safety Rule Approach

Lichtenberg and Zilberman (1988) suggest an approach to policy determination that accounts for uncertainty. It combines a probabilistic risk assessment model with a safety-rule decision mechanism that is equivalent to the use of significance levels for statistical decision making. The resulting estimates of the uncertainty-compensated trade-off between risk and social cost can be used for policy determinations that rely on formal decision criteria like benefit-cost or risk-benefit.

The model of Lichtenberg and Zilberman is consistent with legislation that charges policymakers with balancing cost and protection within a given margin of safety. The model can be expressed formally as follows: Consider a set of M policies. Let $X = (X_1, \ldots, X_M)$ denote the extent to which each policy is used. The cost imposed on society from choosing a set of policies X, denoted $C(X)$, is assumed to be a monotonically increasing function of X: The greater the extent to which any policy is employed, the greater the social cost is assumed to be. Social cost in these cases includes cost of cleanup, government monitoring and enforcement, protective measures taken by the public, and producer and consumer market welfare costs.

The maximum allowable risk and the margin of safety are taken as given in this analysis. The determination of the safety margin is akin to the determination of a confidence level in statistical testing. Determination of the maximum allowable risk is generally a political decision. This model, however, provides useful measures for determining risk standards. Every maximum allowable risk standard implies a shadow price that can be interpreted as society's willingness to pay for marginal risk reductions within a given safety margin. The value of risk reduction decreases as the margin of safety increases, and increases as the maximum allowable risk decreases.

The safety rule can be expressed as a condition specifying that risk, R, be constrained to remain below a given maximum allowable level, R_0, within a given margin of safety, P. That is, R exceeds R_0 no more than a fraction $1 - P$ of the time. This can be written formally as

$$\Pr(R \leq R_0) \geq P \quad \text{or} \quad \Pr(R \geq R_0) \leq 1 - P. \tag{4}$$

The regulatory decision problem can then be expressed as the choice of a set of optimal policies $X_1^*, \ldots, X_M^*$ to minimize the total social cost of meeting the safety rule: Min $C(X)$ subject to (4). The solution to this problem is a set of policies $X_1^*, \ldots, X_M^*$ that can be characterized in terms of the total cost to society of implementing these policies $C(X^*)$ to achieve a risk standard R_0 with a margin of safety P.

For any target level of risk and any margin of safety, the model can be solved for the optimal levels of pesticide use, damage control activities, averting behavior by consumers, and preventative medical treatments.

The optimal solution involves some combination of pollution control, exposure avoidance, and medical treatment. The cost of reaching the target risk level increases with the safety level. The shadow price of meeting the risk target depends on the degree of certainty we impose that the target is being met. It is easy to see that the margin of safety, P, represents society's (or the decision maker's) aversion to uncertainty concerning actual risk. The higher P is, the more averse the decision maker is to uncertainty.

Lichtenberg *et al.* applied this approach to a case of groundwater contamination in California. They found that the stringency of any given policy varies with the margin of safety required. The cost premium imposed by greater aversion to uncertainty may be quite large, and the marginal cost of risk reduction increases significantly as aversion to uncertainty grows. This implies assessments of policies based on average risk will tend to overestimate allowable risk.

In general, pesticide policy seeks to ensure food safety, agricultural worker safety, and environmental safety. Because consumers are prone to ingesting pesticides from residues in food and water, food safety is a continuing concern (Ferrer and Cabral 1995). Fortunately, the concentration of pesticide residues in foods and the frequency with which they occur have decreased substantially in recent years. Pesticide residues can be found in food from crops or animals either treated with pesticides or inadvertently treated with contaminated air or water. Exposure can also occur with consumption of food animals fed with contaminated products.

In addition, those who work in the production, transport, and application of pesticides can be exposed through dermal contact and inhalation. Farm worker safety is the subject of much attention by policymakers because of the frequency with which workers come in contact with pesticides. Agricultural work is one of the most hazardous jobs in the United States because of the risk of pesticide poisoning (Bureau of Labor Statistics 2005). In 1990, it was estimated that 25 million agricultural workers are annually exposed to pesticides around the world (Jeyaratnam 1990). Exposure can result in both acute and chronic health problems. Cancer and neurological and reproductive damage are among the chronic consequences of pesticide poisoning, while rashes, eye irritation, and breathing problems are common acute effects. Pesticides can also affect the nervous and gastrointestinal systems.

Regulations aimed at improving worker safety include requirements that employers provide pesticide safety training and protective equipment to workers, and that firms give notification prior to applications and label-treated crops with signs that list the chemicals used. In addition, worker-safety policy may include re-entry rules that prevent workers from entering fields for a period of time following applications.

Another externality of pesticide use is its impact on the environment. Once a pesticide is applied to or spilled onto soil, it can remain in place, or transfer to the air, surface runoff, or soil-pore water. An understanding of these contamination pathways is necessary to evaluate the impact of pesticides on the environment. Transfer of pesticides to surface runoff during precipitation or irrigation is a major problem associated with nonpoint-source pollutions, such as water contamination. Drift and vaporization of pesticides also cause the chemicals to spread downwind, not only endangering humans, wildlife, and vegetation, but also potentially causing large-scale, long-term damage that can only slowly be reversed at great expense. Pesticide applications also result in indirect effects on ecosystems by reducing local biodiversity and by changing the flow of energy and nutrients. Ecosystem impacts of pesticides can include effects on human health, domestic and wild animal response, effects on natural enemies of pests, crop pollinators, and on soil microorganism response.

The typical policy response to these externalities has been command and control. Pesticides must be approved for use, and their use may be restricted. Once pesticides are determined to cause illness or to harm the environment, they are often canceled. DDT (dichlorodiphenyl trichloroethane), for example, was the first modern chemical pesticide until it was canceled for agricultural uses in the United States following the publication of Rachel Carson's *Silent Spring* (1962), which alleged the chemical caused cancer in humans. It was also considered hazardous to the environment. Other prominent examples of chemical cancelations include EDB (ethylene dibromide), a soil fumigant used against nematodes and other soil insects, canceled in 1983 amid concerns of adverse effects (like cancer) on humans and the environment; methyl bromide, a broad spectrum chemical used to kill insects, weeds, and rodents, phased out in the United States in 2005 because of evidence it depletes the ozone layer; and 2,3,4-trichlorophenoxy, a defoliant used in agriculture and by the U.S. military in "agent orange" during the Vietnam War, canceled for all crop uses except rice in 1970 and later canceled for all agricultural and other uses in 1980 due to its adverse acute and chronic human health effects. We do not intend to question the famous decisions to cancel these pesticides; but, in general, such policies often ignore the economic benefits of pesticide use. While we reserve a full discussion of pesticide policy to a subsequent section, suffice it to say that taxes and re-entry regulations can accomplish risk reduction without eliminating all economic benefits of pesticide use.

As the economic, biological, ecological, and health fields develop better understandings of contamination pathways and risks, policies should become less blunt to attain risk reduction in the most efficient manner. For example, aerial application of pesticides may generate more than twice the aerial drift of ground application (Pimentel and Lehman 1993). Policies should therefore encourage precise applications. In addition, drainage facilities reduce contamination of ground- and surface-water by runoff and percolation of pesticides, and protective clothing significantly reduces farm worker exposure to hazardous chemicals.

Cropper *et al.* (1992) provide an assessment of the regulatory approach of the U.S. Environmental Protection Agency (EPA), which is mandated by law to weigh environmental and health risks against the economic benefits of pesticides when determining which pesticides to cancel. Their conclusions offer some comfort to economists who would advocate cost-benefit analyses. In particular, the probability of cancelation of pesticides suspected of causing cancer is increasing in environmental and health risk and decreasing in economic benefits. EPA regulation departs from rationality, however, in implicitly valuing the life of a pesticide applicator at $35 million and the life of a consumer who may be exposed to pesticide residues at only $60,000. The authors attribute this incongruity to a bias on the part of EPA to reduce larger individual risks. (The risk of exposure to an applicator is 15 times higher than that of a consumer.) Applicators are also a more identifiable segment of the population than consumers, so a bias to help identifiable lives may also be at work. Van Houtven and Cropper (1996) also find evidence that EPA is biased toward minimizing risk for target vulnerable populations, like farm workers, rather than the U.S. population at large. Cropper *et al.* point out that U.S. law regarding chemical pest control agents requires consideration of economic benefits from

chemicals. However, other laws relating to health and environmental protection preclude cost-benefit analyses in a precautionary approach that demands arbitrarily small levels of risk regardless of cost. For example, the ambient standard-setting provisions of the U.S. Clean Air Act preclude consideration of costs. The effluent standard-setting provisions of the U.S. Clean Water Act, however, mandate consideration of costs and preclude consideration of benefits. In contrast, the U.S. Federal Insecticide, Fungicide and Rodenticide Act (FIFRA) and the U.S. Toxic Substance Control Act (TSCA) require balancing of costs and benefits. There is evidence that such balancing of costs and benefits does influence regulation because implementation of laws by federal agencies may not strictly follow the intent of lawmakers. (The discretion of administrators has been challenged in court.) Even when balancing occurs, as in FIFRA and TSCA, the implicit value placed on a statistical life is \$45 million (in 1989 dollars) (Van Houtven and Cropper 1996).

A growing literature examines biases in the decisions of risk regulators in favor of natural chemicals versus synthetic chemicals; overestimation of risks associated with highly publicized, low probability events; biases in favor of specific constituencies based on identity, location, representation, and activism; and others. Viscusi and Hamilton (1999) offer a nice overview of the literature to motivate their analysis of EPA decision making in the cleanup of hazardous waste sites.

Harper and Zilberman (1992) examine the trade-offs between economic benefits and worker health safety with uncertainty, using the cotton pesticide chlordimeform as an illustration. They compare a safety-rule analysis that protects individuals from excessive health risks to uncertainty-adjusted cost-benefit analysis, which evaluates aggregate trade-offs between health and economic welfare. They find that these two criteria may lead to opposite policy conclusions. They then suggest use of a safe minimum standard, which allows weighing of costs and benefits only after some minimum acceptable level of health safety has been assured.

The external costs of pesticide use, such as environmental damage and acute and chronic human illness, can be substantial. An interdisciplinary effort on the part of economists, biologists, and those in the medical profession is improving our understanding of the risk-generation process and has led to the development of policies that reduce risk without eliminating all economic benefits from pesticide use. The use of taxes and standards governing pesticide residues on foods, farm worker re-entry to treated fields, and protective clothing are less blunt than the traditional response of regulators to cancel pesticides with adverse effects. Complicating the determination of external costs is the heterogeneity that characterizes damage to human and environmental health. This heterogeneity augurs for local responses to such externalities.

5 RESISTANCE

Further complicating the determination of optimal pesticide use is the resistance of pests to chemical insecticides and herbicides. Resistance buildup is a costly problem for the agricultural community that figures prominently in the debates over adoption of GM

crops. It has important implications for environmental and human health. In the United States alone, pesticide resistance costs have been estimated to be nearly $10 billion per year in the form of additional pesticide applications and crop loss from resistant pests (Palumbi 2001). A challenge for economics is to provide sophisticated modeling to more accurately measure these costs. More than 700 plant, insect, and animal species are immune to the chemical agents that once controlled them (FAO 1996). Resistance renders pesticides less effective in preventing crop damage, which diminishes output and prompts many farmers to add ever-stronger doses of chemical pesticides to crops. Fear that widespread adoption of GM crops that include the naturally occurring Bt toxin will spawn pest resistance to the broad-spectrum pesticide has led many to oppose Bt crop adoption. Resistance to Bt would render useless a powerful, but mostly safe, weapon against a host of pests.

Resistance results from a genetic selection process whereby those pests naturally adapted to a control agent reproduce preferentially, changing the genetic composition of the pest population. Specifically, each control agent, be it chemical, biological, or mechanical, operates through a specific genetic pathway. Genetic variation within pest species means some pests within each species are naturally less susceptible to specific control agents. They survive attempts at control whereas the susceptible population is killed off. Survivors, naturally adapted to specific control mechanisms, reproduce in greater numbers, changing the genetic makeup of the pest population by eliminating the susceptible genes from the population. Pest susceptibility is defined as the ability to control pest populations with given control agents.

It usually takes only a decade for insects to develop resistance, and 10–25 years for plant species to resist herbicides (Palumbi 2001). The selection pressure is strengthened with high-dose and frequent pesticide applications. This result is intuitive because stronger and more frequent chemical applications provide less opportunity for those pests containing the susceptible genetics to survive and reproduce. Rather than increasing dosages and applications, the optimal response to resistance is a reduction in the use of chemicals.

Economics has provided an analysis of pest susceptibility as a degradable and nonrenewable natural resource that markets will fail to protect without appropriate intervention (Hueth and Regev 1974). The public good nature of pest susceptibility, combined with farmer myopia, will generally mean pest susceptibility is underprotected. Drawing on the methods of Hueth and Regev, Laxminarayan and Simpson (2002), and Alix and Zilberman (2003), the farmer optimizes profits facing two constraints: resistance growth and pest population growth. The former is increasing in pesticide use, while the latter is decreasing. The first-order conditions for this maximization problem lead to the optimum condition that the sum of the marginal benefit of reducing damage in period t, the marginal product value of control (VMP_{Xt}), and the marginal value of future benefits of reduced pest population (VMF_t), are equal to the marginal private cost of pesticide application (w) and the marginal social cost of resistance from the pesticide application (VR_t). Thus,

$$\text{VMP}_{Xt} + \text{VMF}_t = w + \text{VR}_t.$$

A myopic farmer will not consider the long-term effects of his pesticide use and will instead maximize profits by equating the marginal product of pesticide applications with marginal cost, *w*. This result implies that the myopic farmer may either overuse or underuse pesticides because he ignores both the future benefit of slowing pest population growth and the cost of resistance buildup. Overuse of pesticides relative to the social optimum will occur when resistance costs outweigh the benefits of reducing pest population growth.

Because pests can travel across farms, pest susceptibility is a common resource pool that may suffer the tragedy of the commons. As Alix and Zilberman (2003) point out, this means large farms are more efficient at pest control because they are more likely to contain the movement of pest populations, which internalizes the externalities, good and bad, of pesticide use. Where pest populations do migrate across small farms, collective action is needed to effectively address pest damage and resistance buildup. Such action may be needed to increase pesticide applications if pest control has a positive externality by benefiting nearby farmers who share a pest population.

In addition to the externalities associated with pesticides, information constraints can cause farmers to use a suboptimal level of pesticides. Resistance buildup is not easily detected and is typically not detected during its initial onset. Once resistance is detected, the information is not quickly disseminated to farmers.

Production decisions of manufacturers are, in some respects, a more fundamental question than farm-level decisions because pesticide manufacturers often set prices and use requirements, and they develop new products that clearly impact farmers. Amid sizable external costs from resistance buildup and under conditions of inelastic demand, a monopolist's production may be closer to the social optimum than the production of competitive firms. A monopolist internalizes the resistance externality and considers the long-term effectiveness of its product in its profit-maximizing output decisions.

Because intellectual property rights (IPR) afford developers of new pesticides monopoly power for the duration of patents, we can model the pesticide manufacturer as a monopolist whose profits depend upon product effectiveness, which is clearly diminished by resistance. The extent to which a firm enjoys monopoly power is not just dependent upon patent policy, but is also determined by the availability of substitutes, be they chemical, biological, or mechanical. The monopolist at time *t* maximizes profits by producing where the current period marginal revenue is equal to the current period marginal costs plus marginal resistance cost, where marginal resistance cost is a function of the rate of resistance buildup and of recovery of the susceptible pest population (Alix and Zilberman 2003). (Recall that farmers will use a pesticide in increasing dosage and with increasing frequency amid resistance buildup until alternatives become less costly, at which time they will switch to an alternative control agent. We assume that over the relevant time horizon, as resistance develops, temporary increases in sales will be offset by investment on the part of the monopolist in new chemicals to replace the product with resistance or by profits lost to competitors' substitutes.) The monopolist will underproduce relative to the social optimum unless society's marginal cost of resistance buildup is greater than the resistance cost recognized by the monopolist. The

competitive producer will undervalue marginal resistance cost because of the tragedy of the commons: Each competitive producer will only recognize a share of the resistance cost whereas the monopolist internalizes the whole cost.

An important consequence of this result is that monopolists will invest in resistance management to protect the effectiveness of their products. This effort is reflected not just in output decisions, but also in resistance-related research undertaken by manufacturers and in attempts to solve the common resource problem facing farmers. These companies conduct resistance research and make significant and increasing investments in research and farmer education to manage resistance (Thompson 1997). The average investment of $15 million for each pesticide that makes it to the market provides an incentive for monopolists to protect the effectiveness of their products.

The foregoing analysis suggests that IPRs are important in determining whether a firm will ensure product effectiveness. In particular, theory predicts that once a patent expires and many firms can begin producing generic forms of the previously patented chemical, monopolists will care less about resistance buildup. In a competitive market with the brand-name product and generic substitutes, production will rise above the socially optimal level, and the resistance management activities of the brand-name manufacturer will decline. The brand-name manufacturer will, however, continue to have incentives to deter resistance buildup because it will still enjoy a significant market share due to the brand name and because the company reputation is at stake (Alix and Zilberman 2003).

When resistance buildup occurs, the marginal productivity of pesticides declines, prompting farmers to compensate by increasing dosage strength and application frequency until more productive pesticides are available. The farmer response increases the resistance stock by reducing the population of susceptible pests.

Selection pressure, defined as the force that induces preferential reproduction of certain traits, can be reduced by immigrating pests susceptible to pesticides into areas treated with those pesticides. This is typically accomplished with refuge, which, as we will discuss later in this section, is a stock of land adjacent to treated crops on which pesticides are not used and susceptible pests reproduce without selection pressure. The susceptible pests permit the transfer of susceptible genes to subsequent generations of pests, delaying the onset of resistance. This principle provides the foundation for Bt crop resistance control strategies, as we will discuss later in this section.

Pressure can also be reduced by using a mixture of pest control methods, including chemicals and biological and mechanical control, because alternative damage-control agents act upon their targets in different ways. Because a pest may be susceptible to one agent, but not the others, the risk of eliminating susceptibility is reduced with the use of a mixture of agents or with rotating applications of several agents (Diggle *et al.* 2003).

The use of predators of pests is one way to control damage while also combating resistance. It may be the case that one pest species is predator to another pest species, as in the case of the multi-colored Asian Lady Beetle, which preys on aphids. In this case, a higher level of the predator pest should be tolerated relative to when there is no predator–prey relationship between two pest populations. When the two populations are not independent, the marginal cost of control of the predator pest must include the

marginal cost of a larger prey–pest population. Control of pests that are also predators of other pests should recognize their benefits and cause application levels to be reduced accordingly.

Some of the policies implemented to regulate pesticide externalities have the perverse effect of worsening resistance. The field-entry regulations that we previously considered in our discussion of farm worker safety provide an incentive to use preventive pesticide applications rather than responsive applications, which are made only if pest damage occurs (Lichtenberg *et al.* 1993). Responsive applications could find the farmer having to apply pesticides during a labor-intensive period of production, such as harvest. The regulation would delay farm worker entry, perhaps at significant cost to the farmer. Therefore, the farmer will apply pesticides before the damage threshold is met to avoid applications near harvest. Recall that preventive applications are associated with increased pesticide-use relative to responsive applications, which suggests field-entry regulations worsen the resistance problem.

Similarly, government restrictions on pesticide use and outright bans on certain pesticides can speed resistance buildup by restricting the stock of alternative damage control agents available to farmers. Apart from the effect on resistance already mentioned, there is evidence that these policies, common in the U.S. regulatory regime, have replaced pesticides that act on target pests through several pathways with narrowly focused chemicals that are more susceptible to resistance buildup. As we will discuss later in this chapter, taxes can be used to reduce the quantity of pesticides demanded without impeding resistance mitigation efforts.

Policymakers have centered their attention in recent years on delaying resistance to Bt crops, which, unless properly managed, could be rendered ineffective in a matter of years. The resistance problem facing pesticide-resistant crops exceeds that of traditional pesticide applications because the plant genetics are modified to include the gene that produces the toxin. This means the toxin is always present in the crop. Selection pressure, therefore, is greater with Bt crops than with traditional pesticide applications. The selection pressure cannot be reduced by varying damage-control agents or reducing applications. Consequently, countries that have approved GM crops have mandated that farmers plant refuges of non-Bt crops where susceptible pests can survive and interbreed with resistant pests to maintain the stock of susceptibility. There is little consensus on how large the refuges should be to provide ample protection against resistance. This is owed, in part, to the fact that optimal refuge size is a function of the characteristics of the pest population, the region, and the crop. The EPA, which has regulated Bt crops since their inception, has mandated refuge equal to 20 percent of the land devoted to Bt crops. Recommendations from economists and scientists range from 5 to 80 percent, depending on whether supplemental pest control is allowed on the refuge (Hurley *et al.* 2000). Many countries, particularly those in Europe, have banned GM crops out of fear that Bt crops will inevitably lead to resistance of Bt.

Hurley *et al.* provide a model for determining optimal refuge by maximizing agricultural productivity $\left(\prod (\Phi) \right)$ subject to the constraints of keeping pesticide-use and resistance levels at or below politically and ecologically acceptable levels. Let Γ and Ω denote the maximum acceptable levels of pesticide use and resistance, respectively.

Resistance is denoted R_t and $\gamma(\Phi)$ is the level of conventional pesticide use. The share of refuge provided, denoted Φ, satisfies $0 \leq \Phi \leq 1$. The maximization is subject to the evolution of resistance, the evolution of pest pressure, crop damage, and the cost of production. Therefore, optimal agricultural productivity is obtained by solving the Lagrangian:

$$L = \prod(\Phi) + \Lambda_\gamma(\Gamma - \gamma(\Phi)) + \Lambda_\omega(\Omega - R_t),$$

where Λ_γ and Λ_ω represent the shadow values of marginal increases in the maximum allowable levels of pesticide use and resistance, respectively. As the authors point out, there are complex interactions among refuge, the treatment thresholds that determine conventional pesticide use, resistance, and pest pressure.

An important consequence of this model is that increasing refuge may tend to increase selection pressure and therefore risk. This is because more refuge is associated with greater pest pressure, which reduces the threshold for conventional pesticide application, prompting greater levels of conventional pesticide use. Applying the model to the case of Bt corn and the European Corn Borer in the northcentral United States, Hurley *et al.* (2000) conclude that refuge does reduce resistance, but that the marginal cost of reducing resistance is quite high and the marginal benefits depend heavily on population dynamics. (The benefits of refuge are greatest when pests rebound quickly from pesticides; and if they do not, there are few pests left to manage, so the benefits of refuge are low.) The high marginal costs imply that it may be suboptimal to attain an arbitrarily small level of resistance.

This discussion of the optimal size of refuge is critical to the role Bt crops will play in agricultural intensification. Given the demands on land (housing, commercial, and habitat restoration), the need for an ever-greater food supply, and demand for renewable energy, Bt crops may be the key to producing greater yields on less land. However, because Bt technology must be accompanied by refuge, it may not be the panacea many hoped it would be. Instead, it may increase the amount of land devoted to agriculture (Laxminarayan and Simpson 2000). Furthermore, Bt crops often lower the marginal cost of production, which leads to lower prices and may lead to even greater demand for food. This, in turn, would have the effect of increasing the land devoted to agriculture (Laxminarayan and Simpson 2000).

5.1 Conclusion

Pesticide resistance is a considerable challenge to the agricultural community because of the ongoing need for additional productivity gains. The advent of chemical pesticides in the last century enabled per capita food production to increase even as the world population more than doubled. The demands of a growing world population and an energy industry in need of alternative fuels require continued productivity gains. Yet, even a promising driver of productivity growth, GMO, is subject to resistance buildup. This dynamic creates a race between human innovation and pest resistance. Because pest susceptibility is a common resource, it is subject to the tragedy of the commons. There is, therefore, a need for collective action to ensure proper investment in maintaining

pest susceptibility. Theory suggests that in the presence of resistance, large farms and monopolies may lead to pesticide use nearer the social optimum because they internalize the resistance externality.

6 THE ECONOMICS OF PESTICIDE POLICY AND REGULATION

The environmental pollution and human health risks associated with chemical pesticide applications present a strong case for policy intervention to achieve pesticide-use levels acceptable to society. Because the costs of a pesticide application often are not borne exclusively by the farmer or firm that makes the application decision, economic theory suggests that pesticides will be used in excess relative to the level preferred by society. Although the external costs of pesticides receive far more attention, there are external benefits to pesticide use, which may lead farmers to underutilize them relative to the social optimum. Such benefits principally accrue to owners of adjoining farms who may benefit from reduced pest populations. Often, however, policies are needed to reduce pesticide applications.

Effective pesticide policy is made difficult by a variety of characteristics of the externalities, including the multidimensionality and temporal and spatial heterogeneity of damages, the diffuse and nonpoint source nature of pesticide pollution, information asymmetries, and monitoring costs. Over the years, a number of instruments have been proposed to affect pesticide-use levels, though none has been able to overcome these challenges to achieve a first-best solution. We discuss the costs and benefits of these instruments, which include economic instruments and command-and-control regulation, as well as the interaction of pesticide policy with agricultural policy.

6.1 The Goal of Policy

Economic theory suggests the efficient use of pesticides occurs where the (expected) marginal benefit of pesticide use (including future benefits) and the (expected) marginal social cost of pesticide use are equated. If the total benefits of an additional pesticide application exceed the total cost, including application costs and externality costs, then the additional application should be made. Assuming diminishing marginal returns to pesticides, applications should continue until the benefit of the last application is equal to its cost. Throughout this chapter, we have assumed this condition for the socially optimal level of pesticides. We will continue to do so in this section.

It is worth pointing out that actual guidelines of pesticide levels are different from those defined by equating marginal benefit and marginal cost. Some laws demand a precautionary approach, reflecting the public's wish to reduce risk to arbitrarily small levels. Such laws preclude consideration of economic benefits of pesticide use in their zeal to protect against adverse health and environmental consequences. "Safety first" has been the dominant paradigm for pesticide and environmental health regulations. Despite continuous attempts to reform regulations, most policies affecting pesticides and many other environmental health risks display lexicographic patterns, where investigations

and regulations are triggered by discovery of risks. DDT, DBCP, and others were banned once they were discovered to generate significant damage.

The optimality of pesticide levels that equate marginal benefit and marginal cost is not contingent on a risk-neutral public. Rather, the marginal cost of pesticide use should be defined as the loss in social welfare from a pesticide application, and marginal benefit should reflect the gain. The literature on externality policy refers to this condition as the social optimum, and we adopt that convention. The first-best policy is that which achieves the social optimum with no deadweight loss. All policies are defined against the first best, with second-best policies comprising those that minimize deadweight loss.

6.2 Policy Challenges Posed by Pesticide Externalities

Achieving first-best policy solutions to pesticide externalities is generally not feasible because of the particular characteristics of the externalities, including the heterogeneity of pesticide effects on the environment and humans. Like other forms of agricultural pollution, the environmental and human damages imposed by pesticide applications vary by farm, region, and time. In addition, damages are heterogeneous across chemicals and application methods.

To see how damages may vary by these characteristics, consider a farm on sloped land bounded by a lake that houses endangered aquatic life and that is adjacent to a residential community. Another farm is located in a desert, far from any population center. Pesticides applied to the first farm may easily run off the land into the body of water, contaminating the water and further endangering flora and fauna. Pesticides may also drift through the air to endanger the human population in the residential community. Pesticides applied to the second farm are less likely to flow off the farm because the land is not sloped and are less likely to impose significant costs on human or environmental health.

Now suppose two identical farms apply pesticides. The first uses aerial application whereas the second uses precision technology. Damages associated with the first application would likely be greater because the pesticides can more readily drift off the farm. In addition, pesticides applied during the dry season are less likely to impose significant costs on the surrounding environment and human populations than pesticides applied during the rainy season because runoff is less likely without rainfall.

Table 2 lists some of the externalities associated with pesticides and characterizes their impacts as farm-level, regional, and global. Designing corrective policies is much easier when the magnitudes of the externalities are independent of the location of the polluting activities. For example, the impacts of pesticide use on food safety depend on the amount of pesticide residues in the food and the health and behavior of the consumer rather than the location of the application. In addition, some pesticides may impact the global environment but have no specific or varying local effects. For example, the pesticide methyl bromide is believed to cause depletion of the ozone layer, which affects the global environment regardless of where application occurs. There is no localized effect.

Other types of external effects depend significantly on location-specific elements. As we considered above, when a chemical is applied near a city, any human health effects due to groundwater contamination or aerial drift are greater than if the pesticide were applied

Table 2. Sources of suboptimal use of pesticides and their impacts

Source of suboptimality	Level and cause of impact		
	Farm	Region	Global
Pesticide resistance	X	X	
Destruction of beneficial species	X	X	X
Worker safety	X	X	
Food safety			X
Groundwater contamination	X	X	X
Damage to nontarget species	X	X	X
Air pollution	X	X	X
Property damage		X	

away from a population center. Lichtenberg *et al.* (1993) show the damage associated with exposure to sprayed chemicals varies between regions that have dry harvest seasons and wet harvest seasons. Furthermore, the method and timing of application matter.

The heterogeneity in pesticide costs and benefits implies optimal use can be achieved by financial incentives only if they are specific to chemicals, uses, localities, application methods, use levels, etc. The difficulty in designing simple tax systems to address pesticide externalities may justify the use of direct controls in many cases. The wide array of pesticide regulations we witness reflect, to a large extent, the heterogeneity and multiplicity of the impacts of pesticide use. Computational limitations and enforcement and implementation costs have led governments to use direct controls, and in many cases they present the best available regulatory approach. Economic research can, however, be valuable in assessing and improving various regulations.

First-best pesticide policies must be calibrated to account for this heterogeneity. Policies must induce greater pesticide reductions where the costs are greater. The difficulty of achieving these conditions has led to a focus on second-best policy alternatives.

Another obstacle to the implementation of effective pesticide policy is the nonpoint source nature of pesticide pollution. More generally, it is difficult to determine cause and effect in farm pollution. This is due to the complicated environmental fate processes that determine the delivery of pollution to environmental resources. It is also caused by the diffuse nature of the pollution source. In any given region — a watershed for instance — there are many farms that may contribute to the contamination of water. Determining exactly how much environmental pollution is attributable to which farms requires costly monitoring. The monitoring costs are prohibitive, and precise determinations of the origins of pollution are seldom made. With point-source pollution, it is possible to stop delivery of pesticides to the environment, but the diffuse nonpoint nature of farm pollution renders such instruments largely infeasible.

Because it is difficult to assign pesticide pollution to individual sources, one may attempt to infer pollution from the use of agricultural inputs, which may be more

readily observable. However, complicated production processes and uncertainty about how pollution is delivered to the environment render input-based mechanisms imprecise. Therefore, pesticide policy design has focused on development of second-best policies that minimize deadweight loss.

6.3   Economic Instruments

In the presence of externalities, market outcomes will be inefficient. If the actions of one market participant provide a beneficial service to society at large, then the individual may need to be compensated to provide the socially optimal level of the service. If the actions of a market participant cause harm to society, such as pollution, then the individual may need to be charged for those actions to maximize social welfare. In markets with externalities, financial incentives can be used to achieve socially optimal behavior. In the case of pesticide pollution from farms, a first-best policy would impose an excise tax on pesticide pollution equal to the external cost of pesticide pollution at the socially optimal level. Farm pollution, however, is not observable, so taxing farm emissions is infeasible. Furthermore, the external cost of pesticide pollution varies with the spatial and temporal characteristics of pesticide applications. One-size-fits-all taxes that ignore such heterogeneity cannot achieve first-best outcomes.

Given that the pollution of individual farms is unobservable, financial incentives can be, and are, applied to pollution-causing inputs to achieve a second-best outcome. The pollution from pesticides may not be known, but the pesticide use of farms is observable and may be subjected to taxes and subsidies. The downside of input taxes is that there is not a direct and invariant relationship between pesticide use and damage to environmental resources and human health. The environmental fate processes by which pesticides applied to crops are delivered to the environment and impose damage are poorly understood and vary with soil and land conditions, the climate, and other characteristics associated with farms and application technologies.

Even if there were a direct relationship between pesticide-use levels and damage, input taxes could not achieve optimal pesticide use without deadweight loss. Input taxes distort farm-level decisions, leading to a reduction in input use, but not any other abatement effort that may be less costly. Nevertheless, input taxes can be efficient. Inelastic pesticide demand may, however, require taxes so large that they become politically infeasible (Weersink *et al.* 1998).

Taxes can also be applied to agricultural outputs other than pesticide pollution, which is assumed to be unobservable. It is possible to tax or subsidize outputs correlated with pesticide pollution, and such instruments can be efficient. Crops that are factor intensive may be taxed, while those that are environmentally benign may be subsidized. Like input taxes, output taxes also distort farm-level decisions and, therefore, are not first-best responses.

Taxes should be applied so that the size of the tax moves production to the socially optimal level, which may vary depending on when, where, and how they are applied. The deadweight loss of input and output taxes can be minimized by applying taxes only to pesticides with homogeneous effects or by differentiating taxes based on specific circumstances. Local taxes have been proposed as one way of improving efficiency

because the magnitude of taxes can be calibrated to specific characteristics of different regions. Related to input taxes are green payments — subsidies to farms that adopt less-polluting practices. Such payments may be politically feasible because they increase farm income. However, they are not budget balancing. They also create distortions in farm-level decisions.

Liability rules are another form of financial incentive that has been implemented to internalize the pollution externality associated with pesticide use. Such rules hold farmers responsible for damage associated with their pesticide applications. Such rules can either hold farmers responsible for any damage resulting from their pesticide use or hold them liable only if they are negligent in preventing damage. Again, the nonpoint source of pollution and complicated environmental-fate processes render it difficult to determine liability in practice. Liability rules are also process intensive. Damages and liability must be assigned through the legal process, which can be both costly and time consuming. Segerson (1988) has suggested that social planners can induce socially optimal pesticide management by imposing collective punishment in the form of ambient taxes whenever over application of chemicals causes water quality to fall below an acceptable threshold. This analysis implies that draconian punishments should be used as a deterrent and that rational behavior should ensure they are never used.

Xepapadeas (1991) proposes to solve the difficulty in assigning blame by randomly fining at least one among many polluters if environmental conditions exceed an acceptable threshold. Govindasamy *et al.* (1994) proposed to modify the random fine by determining ordinal rankings of farm abatement efforts and assigning fines according to the rankings. This instrument requires some information of firm-level decisions, but does not require the monitoring necessary for precisely determining each farmer's contributions to pollution.

Other economic instruments proposed to reduce pesticide pollution to socially optimal levels include the assignment of property rights, user-fee tournaments, and tradable permits. As Weersink *et al.* (1998) note, farmers may fail to protect wildlife because they do not have property rights to wildlife on privately owned farmland. It is proposed that if farmers could sell the rights to hunt on their land, the provision of wildlife would be nearer socially optimal levels. As it is, farmers have no financial incentive to reduce pesticide applications to protect wildlife. It has also been suggested that restrictions on the quantity of pesticide use can be efficiently implemented by a system of tradable permits for pesticides over a given jurisdiction, defined, for instance, by watersheds. Development of permit trading systems can lead to efficient use by those farmers who value the limited quota of pesticides the most, but they must also recognize the heterogeneity in costs of pesticide use across space and time. Permit trading has yet to be implemented for the control of pesticides.

6.4   Pesticide Regulation

In light of the difficulties surrounding assessment of blame for environmental pollution in the context of pesticide pollution, and given the heterogeneity of effects, many governments have preferred command-and-control mechanisms to reduce pesticide use.

Pesticide regulation is largely a two-part process that begins with *ex ante* testing of chemicals before they are permitted for use. The second part of the regulation process entails enforcement of registration requirements and ongoing monitoring for side effects from pesticide use. Typical command-and-control mechanisms include bans, restrictions on use, and limits on the total quantity of chemical used. Use restrictions can regulate when, where, and how pesticides are applied.

The current system of pesticide regulations in most OECD countries includes a very elaborate system of testing and analysis prior to the introduction of new pesticides, continuous re-evaluation and re-registration of existing materials (undertaken both routinely and in response to crisis situations), and regulation of application procedures and residues (OECD 1995). It mirrors the system used to regulate pharmaceuticals in the medical industry. U.S. policy, in particular, is reliant upon registration of new pesticides and cancelation of those in use that are determined to have deleterious effects on environmental and human health. European regimes are characterized by a greater emphasis on market-based approaches than those of the United States, although Massachusetts and California have adopted pesticide-use taxes.

Pesticide regulation may depart from first-best policies because the regulation process is subject to political–economic factors. For instance, Cropper *et al.* (1992) find that intervention on the part of farm and agribusiness lobbies lowers the probability that carcinogenic pesticides will be canceled. On the other hand, action on the part of environmental lobbies increases the likelihood of cancelations. In addition, an alliance of manufacturers of existing products, environmentalists, and even farmers may seek to restrict entry of new products and erect trade barriers by increasing the stringency of the registration process. This effect may lead to less competition and fewer introductions of new products than is optimal.

6.5   The Effects of Pesticide Bans

The impacts of pesticide, herbicide, and fungicide cancelations depend on the available alternatives. Throughout the agriculture industry, there is a lack of substitutes for existing chemicals, and growers in many areas have become dependent on only one or two chemicals with no substitutes in the pipeline. The lack of substitutes imposes a major obstacle to regulation of existing chemicals because the cost of bans can mean lower yields, higher damage control costs, and higher commodity prices.

One can employ a general equilibrium approach to determine the impact of a ban on equilibrium prices and output, taking into account the interaction of supply and demand and any feedback effects from related markets. Such analysis may also offer an assessment of equity effects by computing welfare changes for various groups. As a result of a pesticide ban, marginal cost of production per acre increases, output declines, and output price increases. The magnitude of the change in output price depends on the elasticity of demand and any feedback effects from related markets, such as markets for substitute goods. The changes in producer welfare are heterogeneous, with welfare of nonpesticide-using farmers increasing due to the increase in output price and the welfare of pesticide-using farmers decreasing if demand is elastic. Consumer welfare may decrease due

to price increases or increase if improved health effects are sufficient to offset price increases.

Lichtenberg *et al.* (1988) study the impact of canceling ethyl parathion, one of the many pesticides used in California lettuce production. They obtain and aggregate the yield and cost effects of the cancelation to estimate the effect on producer supply. These results are then incorporated into a system of supply and demand equations representing the forces that shape market outcomes. Solution of these equations approximates the impact of the cancelation. They find that redistribution of income among producers becomes the dominant effect of the cancelation when supply elasticities are high and demand elasticities are low. Producer surplus of nonusers of parathion increase by one-fourth to one-third the losses of users. Gains and losses from the cancelation are also found to vary by region within the United States because of heterogeneity in producer reliance on parathion.

Zilberman *et al.* (1991) estimate the potential impacts on five fruit and vegetable crops of Proposition 128 (popularly known as "Big Green"). Proposition 128 is a bond and initiative statute defeated on the California ballot in November, 1990, that would have phased out food-use pesticides known to cause cancer or reproductive damage. When a group of chemicals is banned, pesticide substitution possibilities and yields are reduced more than when just a single chemical is banned. The authors find that consumers would have borne most of the cost of the proposition. The expected value of consumer loss is about 25 percent of current expenditure on the five crops. Estimated average impacts on producers vary among crops, but the average expected loss is 0.6 percent of crop revenue, which can be significant given the narrow profit margins many farmers face.

These results suggest that pesticide bans are suboptimal and may actually benefit producers through higher output prices and harm consumers, particularly poor consumers (Zilberman *et al.* 1991). There is also some evidence that pesticide bans, though intended to improve health outcomes, may actually have a negative effect on health because they adversely impact the availability of food for the poor (Cash 2003).

Knutson *et al.* (1990) estimate the economic impacts of complete pesticide bans on eight major U.S.-produced commodities. The estimated cost increases and yield losses are substantial but, because of increased planting and land-use pattern shifts, the output reductions are relatively small. The aggregate net income of the agricultural sector is predicted to change drastically. Because of price effects, the income of the crop sector is predicted to increase by 18 percent, but due to higher feed costs the income from the livestock and poultry-producing sectors is predicted to decline by 27 percent. Because of the ban, consumers are estimated to have an $18 billion annual loss, which translates to a $90 annual increase in food costs per consumer — a 6.5 percent increase in food expenditures for the average U.S. consumer. The relative impact on those with lower incomes is much higher.

6.6　New Pesticide Control Methods

As the risk–generation process is better understood, a variety of new risk control measures are being implemented. For instance, as was discussed earlier in this section, food safety

problems are less local than other effects, and uniform policies may be more efficient. It may be best to impose regulation based on residue levels in food rather than on pesticide use because the same level of pesticide use may have varying effects on food safety depending on postharvest treatment, handling, etc. The European Union chose to set maximum residue levels in its directives 76/895 (fruit and vegetables) and in 86/362 (cereals).

The development of market differentiation schemes based on pest–control treatments and residues would allow consumer choice. When such markets are established, growers will receive differentiated prices according to residue levels, and that will provide signals to growers when they make their pesticide-use choices.

The transfer of responsibility for application of pesticides to licensed professionals constitutes another attempt at reaching first-best solutions. These professionals provide both diagnosis and cure to pest problems, are educated and informed about all aspects of pesticide use, and can be held liable for certain aspects of mismanagement. The emergence of independent pesticide consultants in California and other U.S. states suggests that such an approach may be valued in the future, at least in regions with intensive agriculture and a good educational infrastructure.

The design of policy has to take into account not only current technological options but also the capacity to induce new innovation. Governments should advance information technologies that make it less costly to monitor pesticide use and make self-reporting requirements politically and economically feasible.

The heavy cost of introducing new chemicals, imposed largely by the emphasis on testing of new chemicals, impedes the innovation process and contributes to market concentration. The impediments to innovation can be costly because new technologies may be cheaper to produce and can provide substitutes for chemicals currently in use, thus reducing the opportunity cost of pesticide-use restrictions and cancelations. Governments may need to invest in research because private firms may not.

Social planners should also recognize the path dependence of agricultural practices and consider lock-in effects of the policies they implement. Pesticide policies influence farm decisions that can impact surrounding ecosystems. Such impacts can only slowly be undone, if they can be undone at all.

As an example, consider the Cote d'Ivoire region of Africa, where the government began giving away chemical pesticides to expand cotton production in 1960. The policy worked, and cotton became the dominant crop. The increase in cotton production reduced biological diversity, and the increase in pesticide use degraded the environment, making the region more susceptible to pest damage (Ajayi 2000). The increased pest pressure continues to make the use of pesticides economical as the gap between potential yield of treated crops and those of untreated crops grows (Waibel and Fleischer 1997). While direct subsidies of pesticides have been stopped, credit and loan programs for chemicals and pesticide application equipment continue to provide incentives for pesticide use. Such subsidies, combined with a general lack of awareness among farmers of alternative damage-control technologies, biased the technological innovation process in favor of pesticides (Knight and Norton 1989). This bias prompts Ajayi to conclude that

even superiority of alternative control methods is not sufficient for farmers in the Cote d'Ivoire to adopt them.

6.7   The Interaction of Farm Policy and Pesticide Policy

So far, we have considered a variety of regulatory responses aimed at achieving socially optimal pesticide use and the complexity associated with achieving first-best outcomes. Further complicating the development of optimal pesticide policies is the impact agricultural policies have on farmer incentives and pesticide-use decisions. The procedures that are established to assess pesticide regulations, such as pesticide bans under the assumption of competitive equilibrium, have to be totally modified to address situations where agriculture is regulated.

Consider the impact of introducing a new chemical that affects the supply of agricultural products under two assumptions. First, there is competition and no intervention in the output market; and, second, output price is controlled by the government and producers are assured a target price. Under competition with no intervention, increased supply will lead to lower output price and increased consumption. Consumers will benefit by paying less for their output and consuming more. The impact on producers is, however, ambiguous. They benefit because the cost of production may decline and output may increase. The reduction in output price may, however, cause them losses, in particular, when the demand for the product is inelastic. Overall, the sum of the impact on consumers and producers is positive and, at least from the commodity market, society is better off because of the introduction of the new chemical.

If producers are assured a target price, however, the welfare effects change. Assuming the target price is greater than the market price, the introduction of a new technology will increase supply, and producers will produce more. However, producers will consider the target price as their supply price, and their production will be according to the target price and not according to the output price. Therefore, they will produce higher volumes than under competition. Consumers will face a price lower than the competitive price because of increased supply. Thus, consumers benefit from the introduction of the new chemical under regulation. In this case, producers unequivocally benefit from the introduction of the new chemical because they produce more output, their costs are lower, and their price remains the same. Therefore, both consumers and producers are better off. The social welfare calculus must, however, also consider the cost of government expenditure to farmers in the form of deficiency payments. These costs must be borne by taxpayers. The introduction of the new chemical will increase government expenditures because output increases and output price declines. In this case, government expenditures may increase so much that the introduction of a new pesticide does not increase social welfare.

Lichtenberg and Zilberman (1986b) use this type of argument to demonstrate that, when prices are supported, an analysis that ignores agricultural commodity programs will tend to overestimate the cost of banning the use of a chemical and overestimate the benefits from the introduction of a new chemical. Using U.S. data, they show that the benefits of chemical use may be overestimated 50–60 percent when price supports are ignored. Analysis should consider that output price does not always represent the

social value of a marginal unit of output. Clever analysis is therefore needed to assess the value of a marginal unit of output produced with chemicals. In cases of subsidy and overproduction, the value of a marginal unit of output to society may be negative because the unit of output may not be consumed but remain in government storage, in which case the output only costs taxpayers money in storage costs.

The effects of land diversion programs are ambiguous. Land conservation programs, however, have an unambiguously positive effect on the environment by taking sensitive lands out of production or specifying low-chemical uses. These policies have been pursued throughout Europe. The environmentally sensitive areas designated under the UK Agriculture Act of 1986 provide restrictions on pesticide use under voluntary compensation agreements with farmers. German law prohibits spraying near surface and coastal waters, and both German and Swiss law prohibits use of certain products in water catchment areas (OECD 1995).

Commodity programs may cause overproduction and a negative environmental side effect not only because of the increase in output price and reduction in uncertainties they entail, but also because of the procedures they use for establishing entitlement. Many commodity programs establish base yields and acreage according to past activities, and farmers recognize that their present behavior affects their future entitlements. Thus, even when output prices are low, a farmer may be engaged in highly intensive production practices in order to preserve his yield base. Farmers may also be reluctant to rotate their cropping pattern in order to preserve their yield base. Therefore, policies that allow flexibility in production and do not require rigid production patterns for entitlements are more beneficial to the environment than rigid policies that may motivate farmers to engage in activities that cause environmental damage.

Input subsidies and especially water subsidies have a significant effect on pesticide use, often encouraging overuse because of overproduction relative to the competitive case. Input subsidies also encourage use of technologies (particularly for irrigation) that increase runoff and groundwater percolation (Carpentier and Rainelli 1997; Chisholm and Wynen 1996).

6.8 Conclusion

In general, the wide array of government subsidies that increase agricultural production tends to increase chemical use. Reductions in pesticide use can be affected by moving toward more competitive markets, adopting efficient modes of production, and reducing subsidization of agriculture. Pesticide policy must consider the distortions other farm policy may cause, as well as the heterogeneity of costs and benefits across space, time, chemical, and crop, which we have considered here. With a better understanding of the risk-generation process, policies should be as narrow as possible to attain acceptable levels of safety, as opposed to the blunt tools that include one-size-fits-all registration requirements and pesticide cancelations.

Much of the crop damage that pesticides aim to prevent is caused by invasive species – those not native to a region. Much as economics can inform pesticide policy, so too can it aid the design of policy to minimize and mitigate the effects of species invasions. We turn

next to that topic and will conclude with a discussion of the issues surrounding the most recent damage control tool available to farmers: agricultural biotechnology.

7   INVASIVE SPECIES

Half of the 50,000 plant species in Australia have been introduced to that continent either intentionally or accidentally. In the British Isles, nonnative plant species now outnumber those indigenous to the region. Similarly around the globe, nonnative species are spreading at faster and faster rates, imposing costs on the global economy on the order of $1.4 trillion every year (Pimentel 2002). Despite the increasing rate of invasions, only 10 percent of introduced species will become established, and only 10 percent of those will become pests (Williamson 1996). Regardless, the spread of invasive alien species has altered ecosystems, reduced biodiversity, endangered human health, fouled water sources, destroyed agricultural land, and significantly altered the evolutionary process. These tremendous costs, combined with the fact that an established invasive species can seldom be eliminated, make the control of invasive species one of the most critical issues facing the global community. The problem, as we will discuss, is largely an economic one.

The precipitous increase in the spread of invasive species can largely be attributed to the speed with which people and commodities are transported across continents and countries, as well as the increasing volume of these movements. Trade is considered the primary pathway by which nonnative species are introduced to a region. Other notable driver of species invasions, particularly relevant today, are terrorism and global warming. Global warming will likely increase pest vulnerability and may lead to interregional pest movements.

Invasive species are also intentionally introduced with the intent that they freely spread, as was the case with the introduction of foreign grasses to improve U. S. grazing land. They may also be intentionally introduced with the intent that they remain confined, as is often the case in agriculture. It should be noted that while we will generally consider alien species as causing damage and their invasions as unintentional, it is the case that alien species introductions may generate a great deal of benefit. In fact, in many parts of the world, nutritional needs are primarily met through the cultivation of crops introduced from foreign lands (Hoyt 1992).

Alien species invasions are typically the unintended consequence of market transactions that fail to consider the cost to society of invasive species introductions. But unlike other externalities economists consider, such as pollution, alien species invasions are self-perpetuating. Once an alien species is established, the individual or firm cannot correct behavior to eliminate or reduce the cost of the externality — the invasion cannot be undone. On the other hand, the external cost of pollution, for instance, can be controlled by the firm through its employment of new technology or reduction of output.

Policy responses can generally be categorized as either controlling invasions or preventing them. Examples of the former include attempts to limit the population of invasive species through pesticides and other means of population control, improving ecosystem resilience, and regulating invasive species uses. Prevention efforts include regulations on

ballast water discharges and screening and quarantine of foreign goods at ports of entry. Despite seeming preferable to after-the-fact control of invasions, prevention is often ineffective and costly. It may, for instance, slow the movement of international trade.

7.1   Control and Prevention of Invasions

Governments seek policies that maximize total social welfare subject to many constraints, including the limited budgets of regulatory and enforcement agencies; the profit-maximizing behavior of importers and growers; and the surplus-maximizing behavior of consumers, market-clearing conditions, as well as political, legal, technological, and information constraints. Policies should seek to balance the use of incentives, monitoring, and control along the species-movement process. The potential for rejection of imports prior to shipment, as well as rejection of imports and the imposition of fines at the border of importing countries provide importers with incentives to ensure their shipments are clean. After alien species have entered the importing country, farmers can be subsidized to detect invasions and protect against them. Monitoring can be done overseas and at the border, as well as within country to detect invasions. Control of invasive species includes treatment of shipments overseas and at the border, as well as chemical spraying to control or eradicate populations within the country. Figure 2 illustrates the movement of invasive species from the country of origin through the importation process and highlights the opportunities for intervention throughout the process.

Cost-benefit analyses are typically used to determine which projects should be pursued. A project should be undertaken if expected present value of control or environ-
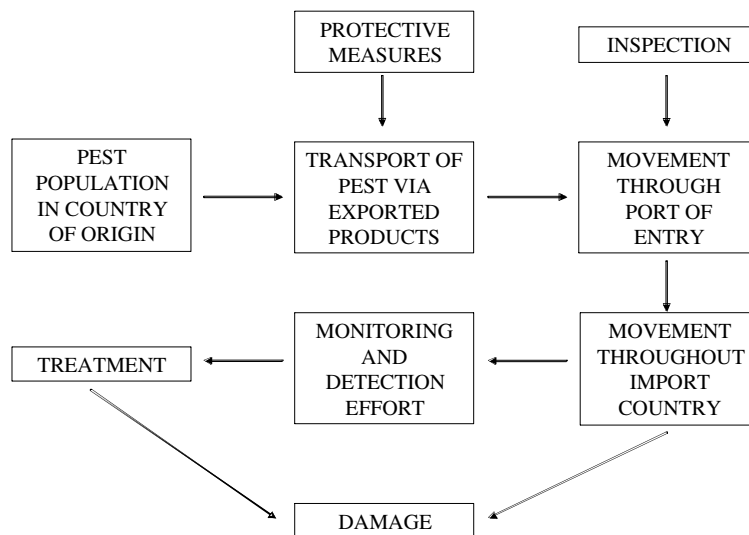

Figure 2. Species population movement process: Driven by biology and economics

mental project exceed or equal the expected present value of the cost of the project. A common response by decision makers to the risk of invasion is to ban imports of any commodity that poses such a risk. As Knowler and Barbier (2005) note, such a response ignores the benefits of imports and invasive species themselves, as well as less blunt policy options that may be available, such as control or financial incentives. As they write, "those invasives that have been deliberately introduced typically have some perceived benefit as a rationale for their introduction in the first place." Thus, the cost of a ban on imports of an invasive species is the sum of foregone benefits to consumers and producers of affected imports and costs directly associated with implementing the policy. These costs, then, should be compared to the expected cost of invasion. As an alternative to a ban, Knowler and Barbier provide a model whereby those who introduce invasive species pay a tax that aligns private analysis with social analysis to produce an optimal level of imports. In a specific application of the model to the introduction of saltcedar (*Tamrisk*) to the United States through nurseries, they find the size of the "introducers pay" tax to be sensitive to the specification of a hazard function. Further, the randomness associated with invasions requires strong assumptions on functional forms that may make imposition of a tax difficult.

As Dalmazzone (2000) notes, the determination of optimal policy response for species invasions is difficult for several reasons, not the least of which is the difficulty of assigning value to nonmarketed environmental services. The challenge of determining the full range of effects of invasions has yet to be overcome in the literature. It is compounded by the diverse effects of invasions on other species and the time-dependent magnitude of effects.

Evaluation is further hindered by the endogeneity of control dynamics. Risk of invasion and cost of invasion, for instance, are functions of human impacts on ecosystems and their efforts to reduce risk. In general, lands altered for agriculture or other uses are more susceptible to invasion and support fast growth of invasive species. Mitigation and adaptation efforts, on the other hand, reduce invasion risk.

Besides the endogeneity of risk, the nature of invasion risk poses additional challenges. The probability of invasion is typically low, but the consequences of invasion are quite costly. A single invasion can be calamitous. Consider that the invasion by yellow-star thistle (*Centauria solstitalis*) of 4 million acres of grassland in California costs nearly $2 billion per year in lost agricultural production. The introduction of neotropical shrub, *Lantana camara*, provides additional breeding ground for the tsetse fly in East Africa, causing higher incidence of sleeping sickness in wild and domesticated animals and humans. And the invasion of 160,000 hectares of the Florida Everglades by the Australian paperback tree (*Melaleuca quinquenervia*) has eliminated natural vegetation and provided poor habitat for a host of native animal species (Campbell 1994; Greathead 1968; Schmitz *et al.* 1997). The power of expected utility is diminished with low probability catastrophic events (Chichilnisky 1998). People treat very unlikely events by either overestimating their probabilities of realization or setting their probabilities to zero. Invasions are also one-time events often independent of history, making estimation of probability density functions impossible (Horan *et al.* 2002).

Finnoff *et al.* (2005a) provide an accessible treatment of these issues and use an endogenous risk framework to model the control efforts of firms, which are assumed to be unable to prevent invasions, and the prevention and control efforts of resource managers (the government). They note that the endogenous risk framework "captures the biological and economic circumstances that jointly determine the level of risk and the optimal mix of prevention and control strategies." In their model, they assume myopia on the part of firms, which follow their private interests to equate the benefits of control with the additional costs of control (where both are assumed to accrue in the current period). The resource manager follows a discount rate based on social preferences and uses collective prevention to lower the probability of invasion and collective control to reduce the damages of invasion. The probability of invasion is assumed to decrease in prevention effort. Damage is a function of the abundance of the invader, which is decreasing in the previous period's control effort.

Following the methods of Finnoff *et al.*, we assume that the manager seeks to maximize social welfare from time t to T. Let W be the total discounted social welfare over the time horizon and let it be defined by θ, a state variable measuring the abundance of invaders at time t. Firms may undertake efforts to adapt to invasions, denoted Z^P, which reduce the magnitude of firm losses and efforts to control the invasion, X^P, which reduce damages. The manager influences θ with collective control, X^G, and prevention, S^G, taking into account the dynamics of the invasions problem. Using a social von Neumann–Morgestern utility function, U, we arrive at the following problem for the resource manager:

$$\underset{\{X^G, S^G\}}{Max} \; W(\theta, t) = U(w(\theta, \hat{Z}^P, \hat{X}^P, X^G, S^G)) + \rho \sum_i \Psi(\theta, \hat{X}^P, X^G, S^G, i) W(i, t+1),$$

where ρ is the discount factor and Ψ is the probability of a change of state from θ to i, given random invasion, stochastic population growth, private prevention, and collective prevention and control chosen to maximize $W(\theta, t)$. Let w be net social benefits for any given state of nature. It is assumed to be a function of firm profits (a function of private adaptation and control costs) and collective control strategies. In essence, the problem is to choose collective prevention and control to maximize net social benefit in the current period plus the cumulative net social welfare in the subsequent period, which is the sum of cumulative net social welfare in each possible state of nature times the probability of each possible state of nature.

The authors use this model to study the effect of considering biological and economic feedbacks on optimal prevention and control efforts as well as the effects of risk aversion on the part of resource managers. In their case study of zebra mussels in a Midwest U.S. lake, they conclude feedbacks do matter and that risk-averse managers tend to reduce prevention effort and increase control. This last result may seem counterintuitive, but as the authors explain, the expected marginal effectiveness of control is greater than that of prevention. Investment in prevention reduces the likelihood of invasion, but does not eliminate it, whereas investment in control certainly reduces the abundance of invaders. The lack of prevention increases the probability of invasion, increasing abundance, adaptation, and control, and lowering discounted cumulative

social welfare. For a more complete discussion of this model and the impact of feed-backs, readers are directed to an article by the same authors in *Ecological Economics* (Finnoff *et al.* 2005b).

In a similar model, Jensen (2002) assumes planners can undertake a flow of expenditures to prevent or delay an invasion until the invasion occurs, at which point they can undertake a flow of damage abatement or invasion control expenditures. Because prevention expenditures reduce the risk of invasion, they delay the need for damage control expenditures and thus reduce those expenditures. Higher prevention expenditure reduces control expenditure. Similarly, lower damage control expenditures reduce the need for prevention. Therefore, prevention expenditure is increasing in damage control expenditure. According to the model, it may be optimal for a country to not invest in prevention and, rather, undertake a flow of expenditures to control the invasion once it occurs. The policies are to be determined so as to maximize utility from net income.

Jensen isolates the effects of uncertainty about the timing of the invasion from those derived from uncertainty about the cost of the invasion. The reader is directed to Jensen (2002) for a full treatment of the model. We will simply state the conclusions of the model. In particular, it may be optimal to make no prevention expenditure even if the range of potential invasion costs is positive. This is a result of the uncertainty of invasion. Similarly, protection expenditures may be optimal even if the expected cost of an invasion is zero.

Once an invasion occurs, reproduction by the invasive species means control efforts may have to be undertaken indefinitely, unless resource managers can successfully erad-icate the species. Olson and Roy (2002) provide a model for when total eradication of an invasive species may be optimal. In particular, let y_t be the size of invasion (measured either in biomass or invasion boundary area) at period t. Let a_t be control effort so that the size of the invasion after control is $x_t = y_t - a_t$. The invasive species then grows according to: $y_{t+1} = \rho_{t+1} f(x_t)$, where ρ_{t+1} is an independent and identically distributed random environmental disturbance and $f(x_t)$ is the biological growth function. Given the cost of control (which includes costs of displaced species, impacts on nontarget species, etc.), denoted $C(a_t)$, a total damage function (including ecological and economic damages), $D(x_t)$, and a discount rate, δ ($0 < \delta < 1$), then the social planner's problem is to minimize:

$$E\left[\sum \delta_t [C(a_t) + D(x_t)] \ \text{ s.t. } \ x_t = \rho_t f(x_{t-1}) - a_t\right].$$

This model captures the fact that invasion is a dynamic process and that control efforts are a function of the rate of invasion spread and the change in control costs over time. If the cost of control grows faster than the damage, it may be sensible to attempt control early on but not continue control efforts indefinitely. On the other hand, if damage grows faster than the cost of control, then control effort may be optimal until it succeeds in controlling the invasion. In the case of invasive species that grows and spreads rapidly but cause low damages, it may be optimal to undertake no control efforts and instead tolerate the invasion.

The optimal level of policy will vary across regions and is a function of the biodiversity at stake, the relative productivity of agricultural land, the nature of invading species, and the size of the invasion. Among post invasion responses, eradication is often the most cost-effective if the invasion is detected early on. Detection of invasions is difficult, however, and typically does not occur until invasive populations have increased substantially. Because of exponential growth rates of invasive species and rates of dispersion, the costs of containment and eradication increase as a function of time elapsed since the invasion (Mack *et al.* 2000). This, therefore, puts a premium on public awareness campaigns and monitoring.

7.2   Invasion Prevention: A Weakest-Link Public Good

Much of the economics of invasive species centers on the public good nature of prevention and control efforts, which predicts underinvestment in their provision. Invasions result from decisions made by individuals who do not internalize all costs of their actions. For instance, prices for agricultural commodities and factors of production typically do not account for the risk their use imposes on society (Perrings *et al.* 2005).

Further compounding this problem, invasion prevention is a weakest-link public good. Benefits to society of prevention effort are determined by the weakest member of society. An invasion is the result of interactions among countries, including a country that is home to a potential invader and other countries connected to it through trade in goods or people. Efforts to contain or eradicate pests will fail even if all members of society do so successfully and one member of society (perhaps a farmer or landowner) does not. This suggests there will be underinvestment in prevention and points to the need for a national policy for prevention and control when invasions occur within borders and an international program for invasive species that cross borders (Perrings *et al.* 2005).

The economic response to these market failures is two-pronged, according to Perrings *et al.* (2002). First, because the problem stems from behavior that does not account for social costs, a system of incentives and disincentives should be developed to induce optimal behavior among actors who are the proximate causes of species invasions. Property rights for natural resources should be established to internalize the externality and reduce the tragedy of the commons. Such property rights should be accompanied by institutions to protect them, including a means of compensating property owners. Perrings *et al.* propose importers of potential invaders be required to purchase insurance in case their imports do cause an invasion. They also suggest an extension of rights adopted in South Africa that permits a landowner to sue a neighbor if a fire started on the neighbor's land imposes damage on the landowner. Such an extension would hold property owners liable for containing invasive species on their land.

To overcome the weakest link aspect of invasions, the authors propose the creation of institutions to support the countries least effective at preventing invasions. They note that the U.S. Centers for Disease Control and Prevention monitor disease worldwide, developing information that is at once useful to the United States and to poorer countries that cannot or do not make similar investments. An international institution to monitor

and report on alien species invasions could perform a similar role and coordinate global responses to what is increasingly a global problem.

While economics can address the market failures posed by species invasions, it is important to realize the role of political institutions in determining responses to the problem. Optimal responses to invasion risk may be to increase trade restrictions, but such policies may not be implemented because of political pressures for trade liberalization. In addition, members of society are typically unwilling to support considerable expenditures on prevention when the probability of invasion is low and the costs of invasion are uncertain. Democratic institutions, therefore, can be a contributing factor to underinvestment in prevention and control.

In the United States for instance, Leung *et al.* (2002) estimate $324,000 to be optimal annual expenditure on the prevention of a single invasion in a single lake. In contrast, the U.S. Fish and Wildlife Service spends $825,000 to control all invasions in all U.S. lakes. Leung *et al.* use a stochastic dynamic programming model to conclude the United States underinvests in prevention.

Economics can help to correct the market failures that lead to alien species invasions. As long as prevention and control are required, however, the determination of invasion costs will be a critical area of research. The damage imposed by invasive species can be far ranging. Invasions reduce potential crop yield, increase damage control costs, increase fire susceptibility, deplete water supplies, transport disease, disrupt fisheries and grazing land, destroy forests, eliminate species, and alter evolution (Mooney 2005). Pimentel *et al.* (2005) estimate that the global cost of invasions is $1.4 trillion per year. Pimentel's estimate of the damage imposed in the United States, $137 billion per year, is higher than the only other comprehensive estimate of U.S. damages, produced for the U.S. Congress. That study measured total losses at $97 billion over 85 years (Pimentel 2002; Office of Technology Assessment 1993). Pimentel explains the discrepancy in saying his analysis is more comprehensive in scope and uses updated costs. However, these two analyses demonstrate the difficulty in reliably estimating such costs.

Pimentel (2002), Pimentel *et al.* (2005), and Huntley (1996) offer interesting case studies of the costs of invasive species. Estimates of damage to specific crops by specific species can be fairly easily quantified. The task of aggregating these costs to determine the total cost of invasive species is more tedious and subject to error. Considering all relevant components of invasive species costs alone is difficult. The summation of the individual components then compounds the measurement error to which each component is subject (Perrings *et al.* 2000).

7.3   Determining Costs of Invasion Impacts Nonmarket Valued Goods

Adding to the difficulty of estimating these costs is the problem of attaching monetary values to biological goods and services for which there is no market. These costs of invasive species have historically been overlooked, perhaps because of this very difficulty. Increasingly, however, attention is being paid to the role invasive species play in diminishing biodiversity. In fact, invasive species are the second-most important reason

for biodiversity loss (Holmes 1998; Mack *et al.* 2000; Lodge 2001). Native species often do not have defenses that allow them to compete with invasive species. As a result, they are either displaced from their habitats or driven to extinction. Of the 256 invertebrate extinctions with known causes worldwide, 109 are attributed to biological invasions (Cox 1993). Invasive species threaten as many as 80 percent of endangered species worldwide, and nearly half of the endangered species in the United States owe that distinction to invasive species (Stein and Flack 1996; Wilcove *et al.* 1998).

The role of invasive species in biodiversity loss is obvious, but how those costs should be measured is not so clear. As Perrings *et al.* explain, the removal of any one species in an ecosystem can cause a huge shock to it. The ability of an ecosystem to adapt to sudden changes is dependent on ecological redundancy, which invasive species diminish. Because ecosystems provide valuable ecological services to humans, such as providing water sources, waste assimilation, soil restoration, nutrients recycling, and flood protection, to name a few, reductions in biodiversity can be costly. Furthermore, such reductions are irreversible — once a species is extinct, it cannot be brought back. Genetic diversity is also of value and is responsible for advancements in the worlds of medicine and agriculture. In addition to these use values, one must also consider the option value of preserving genetic and biodiversity for the future. All of these values are difficult to quantify, which compounds the problem of estimating the costs of invasions.

Mooney and Hobbs (2000) put the costs in perspective in describing the biotic homogenization of the earth. The importance of biotic changes is on the order of climate change, which receives considerably more attention among the press and policymakers, even though, as Mooney and Hobbs contend, the economic costs of biotic changes presently outweigh those of climate change. They further point out that while there is hope climate change can be reversed, there is yet no way to rid an ecosystem of an established invasive species or restore an extinct species.

7.4   Conclusion

The permanency of invasive species impacts puts paramount importance on the development of mechanisms to reduce unintended alien species introductions. Because alien species invasions impose costs not often internalized by those responsible for them, such as importers, there is a need for financial incentives that can align private and social costs. Strong property rights and institutions to protect them, as well as liability rules, can help to internalize the externality. Prevention efforts should reflect the benefit of trade and strive to reduce invasion risk while slowing the global movement of goods and people minimally. The trade-off between prevention and treatment of invasions should also be understood so that efficient policies are instituted. In addition, policies should induce farmers and others to identify invasions quickly, as treatment costs increase with the geometric growth of the invasive species. Finally, because investment in invasive species protection is a weakest-link public good, theory predicts underinvestment in prevention and posits a role for stronger members of the international community to support weaker ones.

8 AGRICULTURAL BIOTECHNOLOGY

Selection for desirable traits and hybridization has been used since the advent of human agriculture. Gregor Mendel's revolutionary experiments on the process of heredity paved the way for modern agriculture and demonstrated that through controlled pollination crosses, characteristics can be inherited in a logical and predicted manner. Since then, many plants have been bred to include desirable traits, including pest resistance.

In recent decades, major advances in plant biotechnology have permitted wider access to genetic sources of plant protection against insects and pathogens. Transgenic plants engineered to contain genes for pest-protection have been field tested since 1988 and grown commercially since 1995. Since then, the commercial planting of transgenic pest-protected plants has dramatically increased.

A large portion of U.S. corn and cotton acreage is planted with transgenic pest-protected plants. Those transgenic pest-protected plants contain genes from the Bt bacterium and produce Bt toxin against different classes of insects depending on the types of Bt toxin. Another major type of transgenic plants, i.e., Roundup Ready, have resistance to the herbicide glyphosate; these traits allow the glyphosate to be metabolized by the crop plant so that it does not affect the crop and can be used to control weeds.

Transgenic pest-protected plants have the potential to substantially improve plant health and agricultural productivity and often lessen the need for chemical pesticide. There are clear human health and environmental benefits that arise from reductions in the application of chemical pesticides. However, there may be potential health and environmental problems associated with transgenic pest-protected plants. Potential health problems include: (1) allergenicity of the product or the introduced gene or genes; (2) unexpected production of a toxic component due to the insertion of the new sequence; (3) expression of a gene product in undesired locations; and (4) antibiotic resistance transferred from GMOs to bacteria in the human gut (Zimmermann and Porceddu 2005).

In addition to potentially negative impacts on human health, transgenic pest-protected crops may also have some adverse impacts on ecosystems in general. Some of the potential negative environmental impacts are: (1) displacement of local biodiversity and elimination of existing forms as new improved genotypes are adopted and old landraces are abandoned; (2) gene flow due to natural out-crossing that may result in the creation of new plagues, especially in the creation of new weeds; (3) harmful effects on nontarget species and waste of biological resources, i.e., impacts on the populations of pollinators, predators, symbionts, natural enemies, etc.; and (4) perturbation of biotic communities and adverse effects over the ecosystem that may result whenever a new life form or genotype is introduced into an environment.

These concerns have led some governments to heavily regulate transgenic crops and, in some cases, ban the technology outright. Europe, in particular, has heavily regulated the import of foods containing GMO and the use of GM crops in agriculture, citing concerns over environmental and human health. In some cases where farmers have planted GM crops, environmental activists have destroyed the crops or intimidated the farmers. Consumer sentiment in Europe is particularly anti-GMO, and it is unclear that

there would be demand for GM crops or food if government restrictions are lifted. In 2005, 56 percent of Swiss voters approved a five-year countrywide ban on GM crops.

The European Food Safety Authority and other European institutions have determined that GM crops are safe to consume and pose only minor environmental risks. The U.S. Food and Drug Administration has also validated the safety of Bt crops used in food production after subjecting them to considerable scrutiny. Given this evidence, some have suggested that European resistance is motivated by the interests of the European agrochemical industry and farmers (Graff and Zilberman 2004). With the bulk of agricultural biotechnology property rights held by U.S. firms, widespread adoption of GM technology would largely benefit U.S. firms at the expense of European firms, which have a comparative advantage in agricultural chemicals. European chemical companies have been able to slow their losses in the global crop protection market, and farmers have been able to differentiate their crops on safety and environmental grounds, maintain subsidies, and secure nontariff trade protections.

The United States, Canada, and Argentina have petitioned the World Trade Organization to review European regulation, alleging an illegal moratorium.

The dominant biotech crops have been commodity crops, i.e., soybean, cotton, canola, and corn, and the dominant traits have been herbicide resistance and insect resistance. Globally, in 2004, growth continued in all four commercial biotech crops. Biotech soybean occupied 48.4 million hectares (60 percent of global biotech area), up from 41.4 million hectares in 2003. Biotech maize was planted on 19.3 million hectares (23 percent of global biotech crop area), up substantially from 15.5 million hectares in 2003. Biotech cotton was grown on 9.0 million hectares (11 percent of global biotech area) compared with 7.2 million hectares in 2003. Biotech canola occupied 4.3 million hectares (6 percent of global biotech area), up from 3.6 million hectares in 2003.

Since 1996, biotech crops have been adopted at an exponential rate in developed and technologically advanced developing countries. In only nine years (1996–2004), the hectarage of biotech crops has increased from 1.7 to 81.0 million hectares, a more than 45-fold increase. In 2004, 5 percent of the 1.5 billion hectares of all global cultivable crop land was occupied by biotech crops.

In 2004, biotech crops were cultivated by 8.25 million farmers in 17 countries, and 14 countries grew 50,000 hectares or more. The United States was home to the greatest percentage of biotech hectarage, with 59 percent, followed by Argentina (20 percent). In developing countries, the adoption of biotech crops has increased steadily — from 14 percent in 1997 to 35 percent in 2004. In fact, the growth rate in recent years is faster in developing countries than in industrialized countries: 35 percent versus 13 percent between 2003 and 2004 (James 2001, 2002, 2003, 2004).

In 2004, the three most populous countries in Asia (China, India, and Indonesia) and the three largest economies in Latin America (Argentina, Brazil, and Mexico) grew biotech crops. However, there remains a large number of developing countries with low technical capacity untouched by innovation. The increasing biotech hectarage in the five principal developing countries (China, India, Argentina, Brazil, and South Africa) is an important trend with implications for the future adoption and acceptance of biotech crops worldwide.

The cultivation of GM crops in some developing countries with high research and extension capacity in biotechnology demonstrates that biotech crops are already making an impact through reduced pesticide costs, reduced risks of poisoning, environmental benefits, and productivity gains. The number of farmers who benefited from biotech crops increased from 3.5 million farmers in 2000 to 8.25 million in 2004. Notably, 90 percent of the beneficiary farmers were resource-poor farmers from developing countries, whose increased incomes from biotech crops contributed to the alleviation of poverty.

Studies on the impacts of adoption of transgenic, pest-protected crops in developed countries show that pesticide use declines. Hubbell *et al.* (2000) find that adoption of Bt cotton in the United States results in a reduction of about two pesticide applications per acre, with most of the reduction occurring in the lower south. They note that Bt adopters not only use less insecticide, but also use proportionally less of the predominant type of cotton pesticide, thereby reducing resistance to those pesticides. Frisvold *et al.* (2003) show that adoption of Bt cotton has reduced pesticide applications in cotton by 60 percent, though the yield effects are on average small (below 10 percent). Falck–Zepeda *et al.* (2000) examine the yield-increasing and pesticide-reducing benefits associated with adoption of Bt cotton in the United States and find that U.S. farmers benefit the most, receiving about 59 percent of the estimated $240 million in benefits per year. Benefits to U.S. consumers are approximately 9 percent, while seed companies and Monsanto, the seed developer, received 5 percent and 21 percent, respectively.

Of all developing countries, China is the most aggressive country in terms of biotechnology research and adoption of GM crops. Currently, Bt cotton is the primary commercial GM crop grown in China. Huang *et al.* (2002) find that in all areas of China, adoption of Bt cotton improved yields and reduced pesticide and labor inputs, thereby increasing farmer income. Moreover, use of Bt cotton had positive environmental and health impacts.

In India, Bt cotton has been perceived by industry and government as a means to reduce pesticide use and increase productivity by combating the American bollworm, a major pest. The results of early trials show yield effects of 30–50 percent and substantial pesticide cost savings (Herring 2003). Qaim and Zilberman (2003) show that trials in 2001 had yield effects of 80 percent, which, with consideration of the high infestation rate in 2001, is consistent with Herring's findings. Similar pesticide reductions and yield improvements are also found in Argentina and South Africa (Traxler *et al.* 2001; Ismael *et al.* 2001).

The benefits of agricultural biotechnology may be particularly significant in developing countries where pest pressure is concentrated. GM crops can be integral in rural development, but restrictions imposed by IPR must be overcome by collaborative efforts to share technologies and lower transaction costs. In addition, the lack of markets in developing countries suggests underinvestment in crop technologies most beneficial for development. Therefore, public investment in research not undertaken by the private sector is important. The preponderance of biotechnology innovations have been made in the private sector, by multinational corporations that can afford to develop, produce, and market the technologies (Zilberman *et al.* 2004). Multinational corporations that

have undertaken the development of GM varieties have accumulated the IPRs to their innovations to protect their investments.

The private sector is expected to underinvest in technologies that target the poor in developing countries. No GM varieties have been developed for several staple crops consumed by the poor, such as wheat and cassava (Cooper *et al.* 2005). Public sector research, therefore, will largely be responsible for the adaptation of technologies to specific local needs. The ability of developing countries to reap the benefits of agricultural biotechnology will be determined by their capacity to undertake such research and development effort. Graff and Zilberman (2001) suggest an IPR clearinghouse can significantly aid developing countries in producing varieties the private sector ignores. In particular, the transaction costs of negotiating the rights to technologies and determining the owners of relevant IPR may be significant for public sector scientists who lack the organizational structure needed for access.

Graff *et al.* (2003) show that 24 percent of biotechnology patents are owned by the public sector and argue that these property rights are sufficient to provide the vast majority of the tools developing countries need to develop their own varieties. Zilberman *et al.* (2004) suggest private sector firms may also be willing to license technologies to developing countries for tax or public relations benefits. These property rights could be comprised of a pool of shared technologies or a "one stop shop" for public sector researchers and greatly enable the poor to capitalize on the benefits of GM plants. Some institutions have begun to develop the framework proposed by Graff and Zilberman, such as Australia's Center for Application of Molecular Biology in Agriculture (CAMBIA) and the Public Sector Intellectual Property Resource for Agriculture (PIPRA), which is the result of collaboration by public research universities (Atkinson *et al.* 2003).

While the present generation of agricultural biotechnology is comprised of pest-resistant crops, the potential for crops that are drought resistant and that tolerate marginal lands and climates are real. Already in development are staple crops that are infused with genes to produce additional nutrients. These developments offer opportunities to reduce malnutrition in poor regions of the world. While the impacts of GM crops should continue to be monitored, the development of the technology should be permitted. Bans on the technology reduce the market and thereby diminish the incentive for research and development of a technology that, though bearing some risk, has yet to demonstrate the considerable environmental and human risks of which its detractors warn.

9   CONCLUSION

The need for continued improvements in agricultural productivity and concern about pesticide resistance and human and environmental safety make determinations of optimal pesticide use a big challenge for economists. The combined pressures of feeding a growing world population and of protecting human lives and natural resources make pesticides an emotional topic and increase the importance of wise pesticide use. This

challenge has and continues to require an interdisciplinary approach to understand the effects of the risk-generation process, the damage function that defines how pests reduce crop yield, the way policies affect the size of pest populations and how population size affects crop damage. The job of economists is complicated by the heterogeneity of costs and benefits of pesticide use, as well as by market failures that often lead to underprotection of the stock of pest susceptibility and the environment. Uncertainty about pest populations and the damage they impose on crops further complicates the determination of efficient pesticide use.

In regulating pesticides, social planners should strive for policies that induce optimal pesticide use through efficient policies, like taxation and subsidization. They should also adopt local policies that recognize the heterogeneity of pesticide use. One-size-fits-all policies will generally be inefficient. Because of uncertainty and heterogeneity, however, command-and-control regimes may be preferable to the extent they reduce transaction costs.

Pesticide policies are determined in a political–economic context that may favor a precautionary approach and certainty to efficient outcomes that bear some risk. They must also be considered in a broader policy context because they are often part of larger policy debates. In particular, agricultural policies can distort input decisions among farmers and must be considered in the development of pesticide regulations for optimal outcomes to occur. Pesticide policies can also be used as barriers to trade to protect domestic industry. European objections to agricultural biotechnology, for instance, may be motivated by concern over the competitiveness of the domestic agricultural sector and a realization that adoption of GM crops largely benefits U.S. firms that hold property rights.

Increasing pressure from invasive species and the development of GM crops are two developments that continue to occupy economists. In the case of both, policy responses are inefficient and need an injection of market-based approaches that can correct market failures. In the case of agricultural biotechnology, the promise of greater yields and reduced pesticide use makes outright bans that deter innovation and limit the market particularly crude. Though there is a role for regulation to be sure, the technology should be permitted to develop as side effects continue to be monitored. Though the future of this technology is far from certain, it may hold the key to solving major challenges we face in the 21st century, including those considered here.

The economics of pesticides is still relatively young. Only now are we beginning to understand the productivity of pesticides, how productivity changes over time, and how environmental factors affect productivity. There is a substantial understanding of the health effects of pesticides, but how knowledge about human health effects should translate into policy is less certain. While this knowledge has translated into policy and innovative regimes have been implemented, the policies have not been evaluated to determine their effectiveness. The literature on environmental effects is weak compared to that of human effects. The effects of pesticides on both domesticated animals and wildlife are not well understood. The majority of research has concentrated on the effects of pesticides on plants, not animals. The economics of pesticides will continue to benefit from collaboration with researchers in the natural sciences. The integration

of biology in economic models of pesticides is an imperative. As we continue to seek a greater understanding of pesticides and its environmental and human health effects, and as we tackle new challenges such as those posed by agricultural biotechnology, scientific knowledge will be critical.

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