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# Productivity and Environmental Trade-Offs of Pesticide Regulation

by

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# Productivity and Environmental Trade-Offs of Pesticide Regulation

When synthetic organic pesticides became available commercially in the years following World War II, they were hailed as a panacea for farm pest problems. Unlike the metallic compounds used earlier, they were relatively non-toxic to vertebrates, giving rise to the notion that farming could take place under "sanitary" conditions. Pesticide use grew rapidly, and pesticides have by now become firmly entrenched in modern agricultural practices. Today, farming in many areas is critically dependent on pesticides. Choices about crop varieties, crop rotations and regional location of production are all heavily influenced by the availability of chemical pest control methods.

But pesticides have not proven to be as safe and simple to use as originally thought. By the early 1960s, the disastrous effects of DDT and other organochlorines on predatory bird populations were becoming widely recognized. Farm production in many areas (e.g., cotton production in the U.S. and Central America during the 1960s) had become enmeshed in a "pesticide treadmill" in which reliance on chemical pest control led to continuously escalating applications. Human health problems from exposure to pesticides have also become a problem. In the U.S., attention has been focused on acute poisonings of field workers and increased risks of long-term effects like cancers, sterility or birth defects from occupational exposures or from residues on produce. In developing countries, there is in addition a high level of concern over health effects from misuse of pesticides and pesticide containers, for example, mistaking sacks of pesticides for flour or reusing pesticide containers to hold food or drinking water.

Decisions about pesticide use thus involve trade-offs: Pesticides enhance agricultural productivity, and thus national income and food consumption, but frequently at a cost of greater harm to human health and environmental quality. Choices about the appropriate uses of pesticides must therefore be made by balancing farm production against human health and the environment. This paper considers some of the key issues involved in making such trade-offs. We begin with a brief survey of pesticide regulation in the U.S. We then provide an overview of what is known about the human health hazards and environmental damage associated with pesticide use. We continue with a discussion of what is known about the impacts of pesticides on agricultural productivity. Finally, we take up a series of conceptual and methodological issues that must be addressed in modeling the trade-offs involved in pesticide policy.

## Pesticide Regulation in the United States

Pesticide use in the United States is regulated by the U.S. Environmental Protection Agency (USEPA) under two statutes: the Federal Insecticide, Rodenticide and Fungicide Act (FIFRA), amended most recently in 1988, and the Federal Food, Drug and Cosmetic Act (FFDCA). FIFRA governs pesticide use directly, while FFDCA affects it indirectly by setting tolerances for pesticide residues on raw and processed foods.

*Registration of Pesticides.* FIFRA requires that pesticides be registered in order to be used. Registration involved labeling every pesticide for (a) the crops it can be used on, (b) the areas (usually states, sometimes counties) in which it can be used for each crop, (c) the specific pests it can be used for on each crop in each area, (d) maximum allowable application rates by pest, crop and area, (e) required safety precautions, and (f) specific restrictions on crop rotations, time of use, etc. This label is an enforceable document laying out legally acceptable conditions of use; use contrary to the label is illegal. Enforcement of label restrictions is delegated to state governments.

FIFRA requires that allowable uses must have benefits that exceed the risks of use to human health and to the environment. To be marketed, pest control chemicals must undergo a registration approval process that includes generation and analysis of data on environmental fate, environmental transport, residue levels, and toxicity, both acute and chronic (cancer, reproductive effects, etc.) in humans and wildlife. If the data indicate negligible toxicity and the chemical has been shown effective against the target pests specified, registration follows. If toxicity is not negligible, then the product will be registered only if the potential benefits -- increased income to consumers and producers of the crops it is to be used on -- outweigh the estimated risks.

Registration (including denial or cancellation of registration) is the main policy instrument USEPA uses in regulating pesticide use. Access to pesticides is extremely widespread, and policing usage adequately would consequently be extremely costly. In fact, neither USEPA nor the states possess sufficient resources to carry out real policing. As a result, USEPA shies away from attempts to fine-tune usage, preferring instead direct control of access to pesticides: Those that USEPA considers acceptably safe are registered and can be sold, while those considered unacceptably unsafe are not registered and cannot be used. FIFRA does give USEPA the ability to recognize variability in risk-benefit balances due to heterogeneity in production conditions because a specific use of a pesticide is defined for a pest on a crop in a given region. Thus, USEPA can and sometimes does register a chemical in some regions (usually states, sometimes counties) but not others.

Pesticides that have been approved for use may have their registration reexamined if further experience indicates that risks to human health or wildlife are greater than previously thought. Accumulation of evidence about risk triggers a process known as special review, which duplicates the registration approval process. Because protocols for environmental safety testing took their current form only in the early 1980s, USEPA is currently engaged in an effort to evaluate all chemicals registered for use earlier in light of current standards.

Cropper et al. have studied econometrically the factors that have historically determined the outcome of special reviews for registered uses of 19 pesticides, amounting to 245 crop/commodity combinations. They find that USEPA engaged in risk-benefit trade-off evaluation at all levels of risk, that is, that continued registration was not automatically assured in cases where risks are negligible (e.g., $10^{-6}$ or less). They find that USEPA placed great weight on risks to applicators, low weight to dietary risk and no weight to risk to mixer-loaders. They also find that USEPA responded to political pressure: Comment from environmental groups increased the likelihood of cancellation while comment from grower groups reduced it. Comments from academics also reduced the likelihood of cancellation.

Registration may also include restrictions on time of use or on other farming activities. For example, it may be illegal to grow certain crops for a year or more after a field has been treated with a persistent chemical, because of concerns about uptake in subsequent years (plant-back restrictions). It may also be illegal for farmers or farmworkers to enter fields for a specified time after treatment because of risks due to residues present (re-entry and pre-harvest intervals).

*Residue Tolerances*. Tolerances for allowable residues on fresh and processed foods are set under section 408 and 409 of FFDCA. In most cases, residue tolerances fall under section 408, which mandates the use of risk-benefit balancing. If, however, pesticide residues concentrate during processing (this is common during drying), then the pesticide is considered a food additive. If it is non-carcinogenic, tolerances are set using risk-benefit balancing under section 408. If it can be shown to cause cancer in humans or animals, it is regulated under section 409, the "Delaney Clause", which mandates that the tolerance be set to zero, i.e., disallows any residues. If the registration of a pesticide is canceled, the tolerance for residues is typically revoked (set to zero) after a period of time set to allow marketing and use of crops in channels of trade at the time of cancellation.

Enforcement of residue tolerances falls under the jurisdiction of the Food and Drug Administration (FDA), which conducts periodic inspections of produce. Shipments that fail to meet residue tolerance may be seized. Imports are particularly subject to scrutiny, and failure to meet residue tolerances may result in increased future surveillance in addition to immediate loss of the shipment.

Many pesticides do not degrade completely by the time crops reach the market. In these situations, residue tolerances effectively set an upper bound on the amount of pesticide that can be applied to the crop and/or the time during the production season that the pesticide can be applied. For example, suppose that the pesticide decays exponentially at an exogenously determined rate $\delta$. Let the application rate be x, the time of application be t and the residue tolerance $r^*$. Then the set of maximum acceptable application rates and times of application $(x,t)$ is defined as the set $\{(x,t) \text{ s.t. } xe^{-\delta t} \leq r^*\}$. In some cases, this set will be a binding constraint on pesticide use; Babcock and Foster analyze one such case.

The case where $r^* = 0$, so that the pesticide must be completely degraded by the time the crop reaches the market, corresponds to what happens when the registration of a pesticide is canceled. USEPA's standard procedure is to revoke all tolerances for the pesticide upon cancellation, effective after time allowed to clear the channels of trade.

## Health Hazards of Pesticide Use

Synthetic organic pesticides were seen as a great advance over the compounds previously available because of their considerably lower mammalian toxicity. Nevertheless, they are substances that are necessarily toxic, and many have been found to be toxic to humans. It is not surprising that mechanisms by which these chemicals adversely affect the target species are sometimes shared by humans and that human exposure can result in acute, chronic or delayed toxic effects. For some chemicals, toxicity in humans is by mechanisms different that those in the target species. In either case, it is generally safe to assume that a sufficient exposure to a pesticide will engender some adverse response in humans. The issue, of course, is what constitutes a toxic exposure? The most often quoted maxim of toxicology is that the dose makes the poison. This is a central point that is often misunderstood by both the public and the media. Chemicals are not like infectious agents in that the body does not make more of the chemical once a few molecules have been inhaled or ingested. The probability of toxic response is, in general, a function of the total amount reaching the site of action in the body. Hence, the first question is not <u>if</u> one has been exposed to a pesticide, or any other toxic chemical, but how much?

It is an unfortunate fact that, even when we know the degree of human exposure to most pesticides, we generally have very little data on the probability that dose will lead to a adverse response, whether it is acute or chronic. Where there is data, it generally relates to acute outcomes like those arising from overexposure to the organophosphate or carbamate pesticides which inhibit the enzyme cholinesterase in the nervous system. There is seldom reliable data on chronic or delayed toxic responses like cancer, and what there is usually comes from epidemiological studies of occupational groups. On the other hand, there is relatively good information about the exposure of workers engaged in mixing, loading and application of pesticides in the agricultural workplace. However, workers are seldom exposed to only a single pesticide and the nature of agricultural work makes it almost impossible to gain a reasonable estimate of the exposure of particular workers over periods of weeks or months, let alone years, which is the relevant time frame in the case of outcomes like cancer.

*Acute effects*. One of the few certainties in the area of health effects of pesticides relates to the issue of acute responses. For example, the organophosphates, as a class, are very acutely toxic in contrast to the chlorinated hydrocarbons like DDT. There are well documented cases of fatal exposures in mixers, loaders and applicators as well as in manufacturing, formulating and transport in the U.S. and probably everywhere else in the world where they are used. Moreover, the organophosphates can present significant risks as residues in the environment. The pediatric literature contains numerous horror stories of children being accidentally exposed in the farm environment and there is a history of residue poisoning among harvesters and irrigators in the arid western regions of the United States (see for example Spear et al. 1977, Spear 1991). Residue poisoning has also occurred shortly after application among various other agricultural groups in the U.S. and elsewhere (Osorio et al. 1991, Spear 1991). An important characteristic of the organophosphates in this regard is that they are readily absorbed through the intact skin, making handling and storage of the technical and formulated material, as well as the disposal of used containers, a very important issue. The organophosphates are worthy of discussion in this regard because they exemplify a class of chemicals for which there can be no doubt that they pose an acute hazard which must be dealt with by government regulation. Such regulations must include mandated educational programs for workers and others in the distribution chain, including firefighters and other emergency response personnel.

Even in the case of acute poisonings due to chemicals like the organophosphates, it is very difficult to find reliable data on the number of cases or on incidence rates whose calculation requires both case numbers and estimates of the number of workers at risk. Medical case reports and anecdotal information abounds, but quantitative estimates of risk are very hard to find. This is because few countries have medical care delivery systems which record in the patient files occupational or environmental factors suspected of contributing to the etiology of the injury or disease. Even rarer are systems for the collection and analysis of such data, even if it was recorded at the site of primary care.

The state of California has made a considerable effort to establish a surveillance system for recording pesticide-related illnesses. This system is operated by the California Department of Food and Agriculture (CDFA) and depends principally on physician reports of illness or injury which, if pesticide involvement is suspected, are mandatory under state law. In addition, the submission of a doctor's report is necessary in order for the physician to receive payment for services rendered in cases of work-related injury or illness. Since it is one of the few such systems, it is of interest to review the information available from that source for the most recent year for which data have been released. In 1988 3,144 reports of suspected pesticide-related injury or illness were reported to CDFA of which 2,848 were deemed to contain sufficient information for evaluation. Of this number 2,118 were judged to have at least a possible link to pesticide exposure. These were further classified as 612 "definitely" related, 755 "probably" related and 741 "possibly" related to pesticide exposure. CDFA reports that of the 2,118 cases, 874 involved agricultural pesticide use and 1,244 non-agricultural use. Interestingly, 666 cases involved residue exposure in fields, on commodities or from structural applications. Of the 2,118 cases, 2,016 occurred in the workplace. Classifying by the type of illness resulted in 991 reports involving systemic illness and 1,127 irritation of the eyes or skin. The general picture that can be drawn from these data, including the total number of cases, has remained fairly constant over the decade of the eighties.

Because of the diversity of California agriculture and the large amounts of pesticide used in the state, the California experience can serve as a guide, at least on a qualitative basis, to what is likely to be happening in jurisdictions without reporting or surveillance systems. The regulation of pesticide use in California is among the most rigorous in the world, so that incidence rates in California are unlikely to be useful in estimating what might be occurring in areas with less strict regulation. The anecdotal evidence tends to confirm, however, that the type of cases that have been seen in California have been seen elsewhere as well; reports of DBCP-related infertility in Costa Rica are a case in point (Weir and Mattheissen 1989).

Very little is known about the economic costs of acute health effects of pesticides. Antle and Pingali estimate a two-equation model in which herbicide and insecticide use affect a farmer's health status, measured by a combination of medical costs plus opportunity costs of lost recuperation time. Health status in turn influences productivity as measured by observed variable production cost. They find that both herbicides and insecticides impair health status. Herbicides had a larger effect than insecticides. Increased illness increased variable production cost substantially, because farmers are forced to substitute hired labor for family labor. At some levels of pesticide use, productivity losses from illness outweigh productivity gains from damage control.

*Chronic effects.* The principal source of human data on chronic effects of pesticide exposure are epidemiological studies, most either focusing on a single chemical in a manufacturing or formulation setting -- where exposure cannot be measured very precisely -- or mortality studies of populations for which there is an almost total lack of exposure information (Stubbs et al. 1984). Evidence derived from these studies is thus, at best, suggestive. For example, a number of studies have reported increased risk for cancers such as malignant lymphoma, leukemia, multiple myeloma, testicular cancer, cancer of the gastrointestinal tract, lung cancer and brain cancer among farmers, agricultural workers and pest control operators in the U.S., Canada, England, Italy, Finland and Germany (for a brief survey see Moses 1989). Occupation is taken as an indicator of increased exposure to pesticides. It is obviously a highly imperfect measure that covers a high level of variability in exposure. In many cases, the increased risk was not statistically significant, a result that could indicate a lack of carcinogenicity in humans or that could simply be due to small sample size. Sharp et al. present a critical survey of the epidemiological evidence on delayed health effects of pesticides, concentrating on exposure to dioxin in phenoxy herbicides, organochlorine insecticides, DBCP and organophosphate insecticides. They find contradictory evidence about the carcinogenicity of dioxin. The studies that showed increased cancer most strongly were the ones for which exposure was the least well measured. In contrast, industrial studies of workers exposed sufficiently to develop chloracne showed no carcinogenic effects. Studies of reproductive effects due to exposure to Agent Orange were similarly contradictory, although there was relatively strong evidence pointing to increased rates of molar pregnancies among exposed Vietnamese women. The epidemiological evidence showed no long term effects from exposure to DDT, although there was some evidence from Brazil that DDT exposure contributed to premature birth. In contrast to the evidence on acute effects, evidence about long term neurotoxicity of organophosphate exposure was inconclusive. In fact, clear-cut results were obtained only in the case of increased male sterility from exposure to DBCP among chemical plant operators.

Pesticides may have long-term effects other than cancers or birth defects. Pesticide use has been associated with numerous sublethal health problems, including neuropathy, hyporeflexia, bronchial asthma, impaired liver and kidney function, chronic gastrointestinal diseases, eye problems and dermatitis(for a survey see Coye 1985, Pingali, Marquez and Palis 1991). In a case control study of the Philippines, Pingali, Marquez and Palis (1991) found significantly higher rates of respiratory, neurological and dermal problems in farmers using pesticides more intensively. Pesticide exposure in countries like the Philippines are probably higher than in the U.S. because application and storage practices are less safe and because contaminated foods (e.g., wildlife from paddies) makes up a larger share of the diet. Overall health status is generally poorer in such countries as well. Thus, sublethal health effects are likely to be more pronounced.

Data on human exposure to pesticides is insufficient to establish linkages between exposure and increased chronic risks qualitatively, let alone provide quantitative estimates of the likely health hazards consequent on alternative pesticide regulations. Yet, if risks and benefits are to be balanced -- as U.S. law requires and standard political wisdom suggests should universally be the case -- quantitative estimates of both risks and benefits are essential. In the absence of good human data, policy makers have been forced to rely on indirect evidence, including toxicity data derived from animal testing and exposure estimates derived from simulation modeling. This reliance on indirect evidence in constructing risk estimates makes uncertainty a central feature of policy making.

The very term risk, in and of itself, implies that human health hazards are subject to uncertainty. Susceptibilities to toxic substances typically vary across populations because of genetic diversity. For an individual drawn at random from an exposed population, then, one can estimate only a probability of an adverse response. Information about exposure is typically limited, and can be refined only to a limited extent during the time frame in which decisions must be made. Limitations on exposure information introduce thus error into the risk estimation process. Chronic health effects have multiple causes and mitigating factors. Many are unobservable, so that science can account for part of observed variations in environmental outcomes. In addition, scientific knowledge is usually limited: Our understanding of the mechanisms of carcinogenesis, teratogenesis and mutagenesis is incomplete theoretically and empirically. Finally, the use of animal data introduces additional uncertainties into risk estimates, because the physiologies or humans and test animals (typically small rodents like mice) are incontrovertibly different.

In sum, the process of risk assessment (that is, estimating human health risks quantitatively) is fraught with uncertainty. This uncertainty may well be the chief distinguishing feature of risk estimates: Estimated standard errors may be an order of magnitude or two greater than the estimated mean risk. This means that policy makers must be concerned with probability distributions of risk, i.e., distributions of incidences of health problems in a given population or of numbers of cases, in making decisions about pesticide use. We explore the implications of this below.

**Environmental Damage from Pesticide Use**

It is well established that pesticides can cause serious harm to various forms of wildlife and to whole ecosystems. Pesticides reach non-target organisms like birds, fish or beneficial invertebrates (bees, worms, predatory or other beneficial insects) through aerial drift at application, through runoff of surface water or through direct contact. Because of their inherent toxicity, they often cause adverse effects.

Probably the best known cases of environmental harm are those associated with the organochlorine insecticides (DDT, DDD, DDE, chlordane, heptachlor, toxaphene, lindane, etc.).. While they are more toxic to invertebrates than vertebrates, they have nonetheless proved to be toxic to birds and small mammals in sufficiently large doses. DDT, DDD, dieldrin, endrin and others have been shown to be lethal to small birds, especially younger ones, and small mammals such as shrews, moles, voles and mice. Wide-area insecticide applications (for example, treatment of forests for spruce budworm control in Maine, treatment for fire ant control in the Southeastern U.S. or treatment for Japanese beetle control in the U.S. Midwest) have been found to increase mortality among small birds and mammals. In addition, because of their persistence and because they accumulate in fatty tissue, organochlorine compounds are subject to the process of biomagnification: Concentrations in animals increase at higher levels of the food chain. High concentrations of these chemicals can be directly lethal. They have also been shown to cause reproductive failure due to eggshell thinning in waterfowl and raptors; in the U.S., for instance, the populations of bald eagles, pelicans, peregrine falcons, cormorants and many others were severely depleted by reproductive failure due eggshell thinning caused by accumulation of DDT residues. (For greater detail see Stickel 1973).

Organochlorines have also been found to be toxic to fish. Biomagnification of residues has tended to be the major mechanism, although concentrations found in water and plankton may be sufficient to cause excess mortality in fry (Holden 1973). High sub-lethal accumulations of organochlorine residues in fish have led at times to bans on fish consumption due to concerns about human health effects (see Bottrell 1979 for details).

Concern over toxicity and reproductive failures from biomagnification of organochlorine compounds eventually led USEPA to cancel the registrations of the entire family and to refuse to register any new products long-lived enough for biomagnification to be a problem. Insecticidal compounds currently in use break down more quickly; to compensate for lower persistence, they tend to be more toxic. Carbamate insecticides, for example, have extremely high acute toxicity. Granular formulations of the carbamate insecticide carbofuran were recently withdrawn from the U.S. market because of concern over kills of small birds and of raptors scavenging on them, and similar concerns have been raised about granular formulations of other carbamate (aldicarb) and organophosphate insecticides (diazinon). Pyrethrum, an natural pesticide, and synthetic pyrethroids are highly toxic to fish, and have been associated with a number of fish kill incidents.

Bees play an important role as pollinators in fruit production as well as producing honey and beeswax. Pesticide use has resulted in bee mortality large enough that the U.S. Congress enacted legislation authorizing compensation for bee losses (Bottrell 1976). In registering chemicals, USEPA routinely places restrictions on pesticide application to avoid pollinator mortality. Pesticide use may affect other organisms that are important in human food production. Pesticide use on rice in the Philippines, for example, reduces populations of frogs, fish and other animals used for food that occur naturally in rice paddies. Damage to these populations reduces the variety of foods produced by the paddy and may have adverse consequences on the composition of the diets of farmers and the rural poor (see for example Pingali, Marquez and Palis 1991).

Accidental releases of pesticides have also resulted in serious damage to wildlife. Examples include the Kepone spill in the James River and lower Chesapeake Bay, which resulted in fish kills and losses to commercial fisheries, and the recent spill in California. While unintended and relatively uncommon, accidents are inevitable in manufacture and transport, and accident-induced damages must be taken into account in formulating pesticide policy.

Finally, it should be noted that pesticides can cause environmental damage in the crop ecosystem, and that pesticide use may disrupt agricultural production in unintended ways. Commonly cited on-farm problems of pesticide use include the development of resistance, target pest resurgence, secondary pest outbreak and species displacement (Bottrell 1979).

Use of pesticides acts as a form of selective pressure on pest populations, killing strains that are highly susceptible and thus increasing the share of resistant strains in the population. Rates at which resistance develops vary across compounds. Resistance to some fungicides has appeared within a year or two, while resistance to some inorganic compounds has appeared only after decades of use. By 1984, for example, pesticide resistance had been reported in 450 species of arthropods, 100 species of plant pathogens and 50 species of weeds (National Research Council 1984).

Pesticides tend to be nonspecific, killing beneficial species as well as pests. Insecticides, for example, kill predatory insects as well as herbivorous pest insects. Prey populations typically recover more rapidly from suppression than predators, since predator population size is more tightly limited by the availability of food. Thus, insecticide application can, after a short respite, exacerbate pest infestation problems.

Insecticide use may also selectively suppress a pest species but not its competitors. With reduced competition, a secondary pest population may become well established and pose problems as severe as the primary pest. An example is the case of pink bollworm on cotton in the Imperial Valley, California. Treatment of cotton for heliothis led to severe problems with pink bollworm. Pesticides are less effective against pink bollworm because it burrows into cotton bolls and thus does not come into contact with chemicals on the surface. Over-reliance on chemical pest control of heliothis thus eventually created a more intractable problem with pink bollworm. Herbicide use can similarly lead to displacement of one weed species population by another that is less susceptible to the herbicides used.

Environmental damage from pesticide use is well-established qualitatively, but deriving quantitative estimates of the extent of damage from alternative pesticide use patterns -- both on and off the farm -- is another matter altogether. Estimating damage quantitatively is subject to many of the same fundamental problems as estimating human health risks. Information on environmental fate and transport is often incomplete, and these processes are subject to random influences such as weather. Susceptibility in a given population is variable, with some individuals having higher susceptibility and some lower. The physiological processes that cause adverse effects may not be well understood at the times decisions must be mad. In contrast to human risk assessment, it is possible to get direct evidence on wildlife toxicity through experimentation, although cost considerations may prevent testing on all potentially affected species. Ecosystems are complex entities, and modeling interactions requires considerable simplification and, thus, error. Measurement error remains a problem as well. In sum, uncertainty is a central element in quantitative environmental damage assessment, just as it is in human risk assessment, and policy makers must deal with it in some way.

**Pesticide Use and Agricultural Productivity**

It is difficult to quantify the contributions of pesticides to agricultural productivity. Agricultural production is influenced by many random factors, including weather and pest infestation levels, so that pesticide productivity is inherently stochastic. Choices about pesticide use are made simultaneously with choices about crop varieties and crop rotations. For example, continuous corn production in the U.S. Corn Belt is largely due to the availability of chemical controls for corn rootworm; in the absence of chemical control, farmers would rely on rotation of corn with soybeans to control corn rootworm. The use of high-yielding hybrid varieties in many developing countries is similarly dependent on chemical pest control. It is thus necessary to account for adjustments in varieties and rotations in making assessments of pesticide productivity. Unfortunately, we rarely (if ever) observe natural controlled experiments involving large scale farming with and without pesticides, making it extremely difficult to estimate pesticide productivity using statistical methods. Instead of hard data, analysts are forced to rely on expert opinion to estimate the direct productivity effects of pesticides. Thus, estimates of the aggregate contributions of pesticides to agricultural production -- like estimates of the health and environmental impacts of pesticides -- are subject to the usual problems inherent in analyzing counterfactual cases: They depend heavily on key assumptions and are subject to considerable uncertainty. This caveat should be borne in mind throughout the following discussion.

One early attempt to provide a picture of the contributions of all pesticides to agricultural production was that of Cramer (1967) who estimated percentage losses of 13 major classes of crops due to insects, diseases and weeds. These estimated losses ranged from a low of 15 percent for rye to 50 percent or more for rice and sugar cane. Losses for wheat, oats, barley, citrus fruit and vegetables were estimated at about one-quarter, while losses for millets, sorghums, corn, potatoes, grapes and oil crops were estimated at about one-third.

Pimentel et al. (1978) estimated that the total dollar value of crop losses in the U.S. in 1974 would have been 42 percent without pesticides, compared to 33 percent with pesticides, implying that pesticides reduced yield losses by 9 percentage points. Insecticides made the largest contribution, reducing yield losses from 18 percent of total value to 13 percent. Fungicides reduced disease losses from 15 percent of total value to 12 percent, while herbicides reduced weed losses from 9 percent of total value to 8 percent. They noted that certain fruit and vegetable crops, including apples, peaches, plums, onions, tomatoes and peanuts were especially dependent on pesticide use and that elimination of all pesticides would probably lead to changes in dietary patterns due to higher prices for these items.

More recently, Knutson et al. (1990) solicited experts' estimates of yield reductions due to the complete elimination of pesticides on major U.S. crops. Sorghum yields were estimated to fall by 20 percent, wheat by 25 percent, barley by 29 percent, corn by 32 percent, soybeans by 37 percent, cotton by 39 percent, rice by 57 percent and peanuts by 70 percent. These estimates have been criticized as excessively high (see for example Ayer and Conklin 1990), and they no doubt are; one problem with using expert opinion is that experts are prone to strategic misrepresentation, attempting to make the strongest case for the policy decision they believe best.

Nevertheless, they are broadly consistent with Cramer's earlier estimates. At the very least, they indicate the importance of chemical pesticides in contemporary agriculture.

We noted earlier that pesticide productivity is inherently stochastic. Damage is limited by potential yield, which is influenced by rainfall, solar radiation and other random factors. Damage is a function of infestation levels, which depend on a complex variety of random factors influencing pest population levels, plant susceptibility, predator/competitor population levels and the like. In addition, the lack of hard data from which to infer pesticide productivity statistically means that estimates must be derived from indirect evidence such as expert opinion. The use of such indirect evidence adds to the uncertainty surrounding estimates of pesticide productivity. It is therefore more accurate to speak of estimated *probability distributions* of losses rather than point estimates of losses.

Rola and Pingali's (1991) survey of estimates of pesticide productivity on rice in Asia gives some sense of the extent of this variability in pesticide productivity. Studies on the Philippines present estimates of crop losses due to insect pests ranging from 16 to 44 percent, while estimates of crop losses due to weeds ranged from 11 to 65 percent. Data from International Rice Research Institute field trials indicate that maximum losses range from 1.5 to 4 times average losses. Zilberman et al. (1991) present some additional evidence. Drawing on five different studies of the impacts of "Big Green", they constructed estimates of average output reductions and upper 95 percent confidence bounds on output reductions for five major fruit and vegetable crops. The upper 95 percent confidence limit estimate was 2-3 times the mean estimate.

## Estimating Impacts of Pesticide Use on National Income

Economists do not view production as an end in itself, but rather as a means of obtaining items of consumption. From this perspective, the contribution of pesticides should be measured in terms of additional income generated rather than in terms of changes in yield alone. Pesticide use can lead to lower food and feed prices and thus greater consumption and greater consumer income. It can lead to greater income -- and thus greater consumption -- for farmers, too, by making farming more profitable.

Relying on yield effects to measure pesticide productivity is misleading for other reasons, as well. Alternative methods of pest control may be as effective as chemicals, but more costly; in such cases, eliminating pesticides would affect producers and consumers even though yields remained unchanged. Moreover, farmers are free to adjust crop production practices, including cropping patterns, choice of varieties, use of fertilizers and labor and so on, to compensate for the absence of pesticides. On a broader scale, additional land may be brought into production to compensate for lower yields. One would thus expect the absence of pesticides to have less of an impact on output than on yields, because land, labor, varieties, crop rotations and other factors can substitute for chemical pesticides.

For example, Knutson et al. (1990) use a simulation model of the markets for major U.S. crops to estimate the effects of eliminating the use of all pesticides in the U.S. The changes in output they estimate are all lower than the estimated changes in yield, because of the substitution of land, labor, fertilizer and other factors for pesticides. Estimates of the percentage reductions in output range from one-fifth the size of the percentage reduction in yield for wheat to three-quarters of the percentage reduction in yield for cotton. In one case, sorghum, output was estimated to increase because of changes in cropping patterns.

Consumers also have the ability to substitute among products. Consumers will substitute among foodstuffs according to relative prices. If oranges are expensive relative to apples, they will buy fewer oranges and more apples, as happens in the fall (apple harvest season) in the U.S. If rice becomes more expensive relative to wheat, they will consume more bread and less rice. Consumers can also substitute between foods and other commodities, such as housing, clothing, entertainment, and so on.

The substitution possibilities of both producers and consumers must be taken into account in measuring pesticide productivity. Economists generally assume that agricultural markets are characterized by four key features: (1) both consumers and producers behave as price-takers (i.e., the industry is perfectly competitive); (2) pesticide use is efficient in the sense of corresponding to the least-cost combination of inputs; (3) consumers and producers possess perfect information; and (4) there is no uncertainty. Under these assumptions, the standard tools of welfare analysis can be used to estimate changes in real consumer and producer income, measured respectively by consumer and producer surplus. Consumer surplus incorporates consumers's substitutions among products, while producer surplus incorporates technical substitution possibilities as well as changes in revenue and cost (see Just, Hueth and Schmitz 1982).

Figure 1 illustrates how such changes in real income are measured. Suppose that a new pesticide is being introduced. The assumption that pesticide use if efficient both before and after the introduction of the new chemical implies that the new chemical induces technological regress, expressed in a lower marginal cost of production because of decreased cost (making pest control less expensive), increased yield or both. Marginal cost (supply) will shift to the right, increasing equilibrium consumption and decreasing the equilibrium price. The change in real consumer income is measured by the change in consumer surplus, given by the area a+b+c. The change in real producer income is measured by the change in producer surplus, e+f-a. It has two parts: a change in revenue and a change in cost of production. If demand for the product is inelastic, revenue will decrease. If this decrease in revenue exceeds the decrease in cost, then producers could actually lose from registering the pesticide. Consumer and producer income are generally assumed to have equal weight in social preferences, so that the change in real national income is taken to equal the sum of the changes in consumer and producer surplus, and equals b+c+e+f.

The welfare costs of canceling a pesticide are simply the reverse of the benefits of registering it. Canceling the pesticide results in technological regression, expressed in a decrease (leftward shift) in supply. Consumers will always lose from cancellation (unless demand is perfectly elastic), but if demand for the product is inelastic, revenue will increase, potentially enough to offset the increase in cost, so that producers could actually gain from canceling the pesticide. Note that canceling the pesticide results in a loss to consumers that is not offset by increased revenue earned by producers; this is termed the deadweight loss of the regulation and is measured by area c+f.

Note that unless demand for the product affected by a cancellation decision is perfectly elastic, the welfare cost of the decision will be shared by consumers and producers, i.e., at least part of the cost of the cancellation will be passed on to consumers. How big the consumers' share is, and how great a share producers pay, depends on the relative elasticities of supply and demand. The more elastic demand is, the lower is the share consumers pay; the more elastic supply is, the greater is the share consumers pay.

Estimating benefits requires knowledge of the parameters characterizing supply and demand. In some cases, sector equilibrium models are available for this purpose. For example, there are several models of the grain-livestock sector or of major crops in the U.S. (Just, Rausser and Zilberman 1990; Osteen and Kuchler 1987; Knutson et al. 1990). If supply and demand functions are not available, the methods of comparative static analysis can be applied using data on equilibrium prices and quantities and elasticities of supply and demand (see for example Lichtenberg, Parker and Zilberman 1988, Zilberman et al. 1991).

**Estimating the Impacts of Pesticide Cancellations: Evidence from the U.S.**

Economists have been studying the effects of pesticide cancellations using the standard tools of economic welfare analysis only within the past few years. Still, some generalizations can be drawn from the few welfare analyses that have been performed.

*The Role Additional Capacity for Supply.* When a country possesses large additional crop production capacity, the contribution of pesticides -- and thus the impact of eliminating them -- may not be too large. For example, the estimates of yield reductions due to eliminating pesticides used by Knutson et al. (1990) extremely high, ranging from 25 to 70 percent (see above). Yet the net impact on consumers was quite small, a $90 per capita increase in food expenditures, corresponding to a 6.5 percent increase for the average consumer. The principal reason for this result is that the U.S. possesses large additional capacity for the production of crops like corn, small grains, soybeans, cotton, rice and peanuts. The initial impact of canceling the pesticide will be to decrease supply and thus increase the equilibrium price. The higher price will make it profitable to bring additional land into production, increasing supply and reducing the price again. If the additional capacity available is large, then, in the long run, the fall in supply can be quite small.

One can contrast the findings of Knutson et al. (1990) for major crops with those obtained by Zilberman et al. (1991), who estimated the effects of California's "Big Green" initiative on five major fruit and vegetable crops (almonds, grapes, lettuce, oranges and strawberries). The expected value of the decrease in real consumer income for these five crops amounted to 25 percent of expenditures, an impact about four times as great as Knutson et al. (1990) found for major crops. The major reason is that production of fruits and vegetables is limited by climate. Citrus fruits cannot be grown in areas subject to freezes. Lettuce production in the winter, early spring and late fall are similarly limited to warmer areas (Florida and the Southwest Desert in winter, the Salinas Valley and other coastal areas of California in the early spring and late fall). Climate gives California a similar comparative advantage for grapes and strawberries.

*Heterogeneity and the Redistribution of Income Among Producers.* When a country is sufficiently heterogeneous in terms of production conditions, the principal impact of pesticides -- introducing them or eliminating them -- may be to redistribute production patterns and thus income among producer groups. In the U.S., for example, production conditions for major crops -- including pest infestation rates -- tend to be quite heterogeneous, varying markedly from on production region to another. The U.S. Corn Belt typically experiences relatively little in the way of insect problems compared to the Southeast, which is hotter and more humid. Eliminating insecticides would increase the comparative advantage of the Corn Belt and lead to increased production there that would compensate in part of decreased production elsewhere. Thus, pesticide regulation may have a significant impact on the *distribution* of income among producers: Growers who rely more heavily on a pesticide or pesticides will likely lose income after a ban on use, while non-users of the pesticide will increase their market share and gain income. In other words, eliminating pesticides (individually or as a whole) alters comparative advantage in favor of areas that are less vulnerable to pest infestation or, in the case of cancellations of individual chemicals, have pest complexes that can be treated more efficiently with alternative chemical or non-chemical methods.

For example, Lichtenberg, Parker and Zilberman (1988) analyzed the impact of canceling the registration of the organophosphate insecticide ethyl parathion on almonds, plums and prunes. Increases in production by non-users of parathion were predicted to make up for a large percentage of decreases in production by former users. Total output was thus predicted to decline relatively little and equilibrium prices were estimated to increase by only a small amount, leading to correspondingly small decreases in real consumer income. Using data from the U.S. plum industry, they showed that when demand is highly inelastic and supply is elastic, the principal effect of canceling a pesticide like parathion will be do redistribute income among producers; as demand becomes more elastic and supply more inelastic, the welfare cost of canceling a pesticide rises.

Similar results were obtained for corn and soybeans in the U.S. by Osteen and Kuchler (1987), who found that canceling the registration of classes of pesticides would result principally in redistribution of production and income across regions. Consumers would lose from cancellation due to higher prices and reduced consumption, but non-users' increases in rent would outweigh users' losses, so that the industry as a whole would be a net gainer.

*The Role of Trade.* Because pesticide regulation alters comparative advantages among producing regions, it will generally lead to changes in trade patterns. If a country is a net exporter of a crop, then regulation that eliminates the use of pesticides will increase the marginal cost of production and reduce exports. If the country's net export demand is inelastic, pesticide regulation may actually benefit the country in the short run: Export revenues will rise and the country will, in essence, export part of the cost of the regulation onto foreign consumers. For example, Lichtenberg, Parker and Zilberman (1988) found that canceling the insecticide parathion would increase export revenue for almonds (a crop for which the U.S. is the dominant world producer) by almost as much as the decrease in domestic consumer surplus. In the long run, however, one would expect competitors to increase their supplies in response to higher world market prices. Thus, in the long run, export revenues could fall.

If the country is a net importer, the effects of pesticide regulation depend on the exact form that regulation takes. Restrictions on pesticide use may increase the comparative advantage of foreign producers and thus, *ceteris paribus*, increase imports. This is especially likely to be true if foreign growers are not subject to the same restrictions as domestic growers are. Baumol and Oates (1988) have pointed out in a broader context that developing countries may well find it advantageous to specialize in more polluting industries as a means of promoting economic growth. In the present context, that means that developing countries might decide to permit continued use of pesticides banned in developed countries as a means of increasing their agricultural exports.

U.S. growers of many vegetable and fruit crops have argued that U.S. pesticide policy is leading precisely in this direction. In their view, the increasingly tight restrictions on pesticide use in the U.S. enacted by USEPA will inevitably increase imports, especially from growers in

Mexico, Central and South America who are subject to less stringent pesticide regulation. However, U.S. pesticide policy consists of residue regulation as well as usage regulation. When use of a pesticide is canceled, the residue tolerance is typically revoked at roughly the same time. The penalties for failing to meet residue tolerances can be quite large for importers, involving loss of the current shipment and tighter future surveillance. Thus, tighter U.S. restrictions on pesticide use will lead to increased imports into the U.S. only in cases where there will be no residues at the time the crop is imported into the U.S. This is unlikely to be true for most vegetable crops, since they typically require treatment for insects and diseases until shortly before harvest. It is also unlikely to be true for most fruit crops, with the exception of soil fumigation, dormant and early season uses and, conceivably, processed foods like orange pulp and frozen concentrate if residues remain only on the skin. For example, growers in Mexico producing for the U.S. market adhere strictly to U.S. pesticide use regulations; growing practices are policed both by the government and by producer federations (Taylor 1991).

We noted earlier that pesticide bans increase the comparative advantage of regions that are less vulnerable to pest infestations or better suited to alternative means of pest control. Latin American countries growing fruits and vegetables for the U.S. market are generally hotter and more humid than the U.S., and thus tend to have worse disease and insect problems. In cases where the residue tolerance is binding, restrictions on pesticide use in the U.S. should increase the comparative advantage of U.S. growers, leading to reduced imports. Only in cases where residue tolerances are non-binding will stricter U.S. pesticide regulation increase imports.

## Conceptual and Methodological Issues in Evaluating Trade-Offs

In the previous section, we saw that estimates of the benefits of pesticide use are generally estimated under the assumptions of perfectly competitive markets characterized by perfect information and no uncertainty. Negative spillovers of pesticide use on human health and the environment are similarly estimated using point estimates, as if information about them is complete and random influences non-existent. But in most cases these assumptions are poor approximations to actual conditions, so that these estimates do not provide a good basis for evaluating risk-benefit trade-offs. In what follows, we consider some of the key ways in which actual conditions deviate from the models typically used and suggest some methodological innovations to incorporate them into trade-off evaluations.

*Pesticide Regulation and the Distribution of Income.* Most analyses treat estimated damage and benefits from pesticide use at an aggregate level, concentrating on efficiency effects (i.e., changes in total national income) of pesticide regulation. But equity considerations are often important in determining policy, so that the distribution of costs and benefits must be evaluated. Distributions of costs and benefits may be markedly different in different situations.

For example, consumers as a group are the chief beneficiaries of a pesticide cancellation when cancellation is undertaken because of concerns about dietary risk due to residues in foods or drinking water supplies (for example, Alar, DBCP, EDB or the EBDC fungicides) or damage to wildlife (for example, DDT and the organochlorine insecticides, granular carbofuran). In cases like these, when a large share of the welfare cost of cancellation is shifted onto consumers, the trade-off between income and environmental quality (including health and safety) is in some sense largely internalized, in that consumers both reap most of the benefit and pay most of the cost.

Producers may be the chief beneficiaries of pesticide cancellations under several circumstances. One is when cancellations are undertaken because of concerns about worker safety, when the pesticide is used primarily by self-employed producers and when those producers lack sufficient information about health risks to make well-informed judgements. In cases like these, when users bear a large share of the cost of cancellation, the trade-off between income and safety is internalized to some extent, in that users both reap the benefit and pay most of the cost.

In other cases, pesticide cancellations imply trade-offs at the level of society as a whole, but not necessarily for individuals. When cancellations are undertaken for worker safety concerns and a large share of the cost is shifted onto consumers, then consumers as a group bear the cost but reap little of the benefit. Similarly, when cancellations are undertaken due to dietary risk or wildlife concerns and the main effect of cancellation is to redistribute production and income across groups of producers, then users of the pesticide bear most of the cost while consumers and non-users reap most of the benefit. Pesticide regulation may also affect the distribution of income in a country. Specifically, one would expect cancellations of pesticides to hurt lower-income consumers the most, because they spend a larger share of their income on food. When pesticides are canceled because of concerns about damage to wildlife, upper income consumers are generally believed to be the principal beneficiaries, because demand for environmental quality is believed to be highly income-elastic. In such cases, pesticide regulation has a regressive impact on the distribution of income. In situations where the costs and benefits of cancellations are not internalized, the distribution of benefits and costs -- how to weight the gains and losses of different groups -- becomes an issue.

*The Role of Agricultural Policies.* Pesticide regulation does not take place is a vacuum; in most countries, agricultural markets are influenced by government policies. In the U.S., for example, subsidy programs such as the deficiency payment and loan programs are major determinants of supply and prices of agricultural commodities. Output subsidy programs are generally believed to give farmers an incentive to increase the intensity of cultivation, by means that include greater use of pesticides than would occur in perfectly competitive markets. As a consequence, the assumption that input use is socially efficient may not be appropriate. For example, a study of the U.S. cotton program of the 1960s by Dixon, Dixon and Miranowski (1973) found that pesticide use could be reduced considerably with little or no impact on output, consumer income or producer income at the aggregate level -- provided that the base acreage requirements of the cotton program were eliminated. In developing countries, governments have subsidized inputs like chemical fertilizers and pesticides as a means of encouraging adoption of high-yielding hybrid varieties. Such subsidies give farmers an incentive to overuse these inputs.

Lichtenberg and Zilberman (1986) analyzed the welfare costs of pesticide cancellations under a pure deficiency payment regime. They argued that, from the point of view of the pesticide regulatory agency, pesticide regulation have a *positive* impact on social welfare in that it reduced the deadweight loss imposed by the agricultural subsidy. Using simulations based on supply and demand parameters characteristic of the U.S. in the mid-1980s, they found that this reduction in deadweight loss could amount to as much as 33-50 percent of the size of the total welfare cost.

Just, Rausser and Zilberman (1990) analyzed the impact of multiple pesticide cancellations on major U.S. agricultural commodities using a detailed structural model of the grain-livestock sector that takes into account supply restrictions in addition to subsidy payments. They assume restrictions on pesticide use that would decrease wheat and feed grain yields by 6 percent annually over a ten-year time period. This implies a total yield reduction of 46 percent, which is somewhat greater than most estimates. They found that the principal effect of these cancellations would be to reduce participation in farm subsidy programs because of the increase in market prices; one of their major policy conclusions, in fact, is that environmental regulations like pesticide cancellations can largely substitute for farm subsidy programs. Increased market prices would cancel out losses due to increased cost, so that farm profits would be unchanged. Livestock producers would suffer large losses, but final consumers would not, while government subsidy payments would fall considerably.

In sum, estimates of the benefits of pesticide regulation must incorporate the effects of other government policies. Doing so from the point of view of economic efficiency is necessary, but not sufficient: At bottom, the issue of how to treat government programs is tightly linked to the question of income distribution, since many government programs are undertaken explicitly to redistribute income among sectors of society. Farm subsidy programs or other forms of government intervention in agriculture provide a case in point, since public intervention in and of itself is generally taken to imply that redistributing income to the agricultural sector is a goal of society as a whole.

*Imperfect Information.* Economists' belief that producers adopt the most efficient (i.e., least-cost) technologies is based on an assumption that producers possess complete information about technological possibilities. For example, farmers are typically assumed to know the productivity of pesticides with complete certainty. Yet there is considerable evidence suggesting that farmers' perceptions of pest infestation levels and potential pest damage often differ considerably from actual situations. Entomologists associated with the Integrated Pest Management movement argued strenuously during the 1970s that farmers overestimated infestation and damage and thus applied pesticides excessively. Empirical studies by Tait (1977) and Mumford (1981, 1982) found that farmers' pest control decisions reflected their individual perceptions rather than actual pest infestation levels. Pingali and Carlson (1985) found that apple growers in North Carolina systematically overestimated pest infestation levels. Rola and Pingali (1991) found that rice farmers in the Philippines had generally accurate perceptions of pest infestation levels, but that their pesticide use decisions did not conform to perceived infestation levels.

To say that farmers act on less than perfect information is not to slight their intelligence. Agriculture is in many ways best understood as the management of crop ecosystems. All ecosystems are complex webs of interdependent relations among species. Some of these relationships are relatively simple to grasp, others are more subtle and less easily understood. In fact, the best in scientific research results in only partial understanding of how crop ecosystems function.

One would expect that farmers with more training, more schooling and more experience -- in short, greater human capital -- would have a more in-depth understanding of how crop ecosystems function and would thus make better management decisions. In fact, empirical studies have found that improvements in human capital lead to improved performance in terms of greater accuracy in perceiving infestation levels and in terms of pesticide use. Both formal schooling and specific training in pest management has been shown to produce lower subjective errors in estimating infestation rates and greater substitution of non-chemical for chemical controls, resulting in lower pest control costs (Pingali and Carlson 1985, Garcia 1989). This suggests that education and training should be thought of as potentially valuable components of pesticide policy. This has certainly been the case in the U.S., where development and dissemination of pest control technologies has been a major function of public institutions like land grant universities and agricultural extension services. One issue in pesticide policy is thus the trade-off between increased funding for research and extension versus restrictions on pesticide use.

*Uncertainty*. We have seen that estimates of damage to human health and the environment from pesticide use are characterized by substantial uncertainty owing to error in estimating risk and to variability in risk across populations that cannot be taken into account in risk estimates. In the U.S., at least, the public appears to be quite sensitive to these types of uncertainty. The work of psychologists indicates that the public perceives as more hazardous effects that have greater uncertainty associated with them (for a summary see Slovic, Fischoff and Lichtenstein 1980). The recent furor over pesticide residues on foods (e.g., Alar on apples) bears this notion out. The best data available for the U.S. suggest that roughly 85 percent of fresh produce in the marketplace have no detectable residues and that almost all of the remaining cases involve residue levels that are extremely small and well below those the EPA considers the maximum safe levels (Food and Drug Administration Pesticide Program 1989). Yet much of the U.S. public believes that pesticide residues on foods pose a serious threat to public health. Policy makers also appear to be quite sensitive to these uncertainties, in part because of public demands for taking uncertainty into account in making regulatory decisions, in part (perhaps) because mistakes are the most visible indicator of poor performance.

The preventive posture of public health agencies suggests an asymmetry in preferences regarding uncertainty: Avoiding false negatives appears to count more than avoiding false positives. This asymmetry is reflected in the posture of the public health profession and in much of the relevant legislation, which requires providing adequate safeguards for public health with a sufficient margin of safety.

Regulatory agencies typically build such a margin of safety into their estimates of human health risk or environmental damage by combining "conservative", or worst-case scenario, parameter estimates in a model that simulates the ways in which pesticides enter the environment, come into contact with humans of wildlife and subsequently cause some form of harm. In statistical terms, the resulting estimate corresponds to an upper limit of a confidence interval (Lichtenberg and Zilberman 1988). Lichtenberg (1991) has pointed out that approach taken by produces estimates whose confidence limits are determined ad hoc: The confidence limit implicit in each estimate is a function of the modeling process rather than of an explicit choice. Each damage estimate has a different, possibly non-quantifiable, margin of safety, making it impossible to compare risk estimates. As a result, it becomes impossible to evaluate trade-offs consistently because the estimated outcomes of alternative policies have different confidence levels and are thus qualitatively non-comparable. This problem can be avoided by using risk estimates that correspond to a predetermined confidence level.

Economists tend to rely on expected utility (e.g., multiattribute decision) or mean-variance models to assess choices under uncertainty. Both require specification of parameters measuring social preferences regarding uncertainty. Obtaining estimates of these parameters is usually impossible. As an alternative, Lichtenberg and Zilberman (1988) proposed using risk estimates that correspond to a predetermined confidence level in an economic optimization framework. Assuming a goal of cost efficiency, they examined the optimal choice of a policy vector $Y$ to minimize social cost $S(Y)$ subject to meeting a constraint on maximum acceptable risk $R_0$ with an adequate confidence level $\alpha$. This constraint can be expressed formally as

$$\max S(Y) \ s.t. \ Pr(R \leq R_0) \geq \alpha. \tag{1}$$

The solution to this problem is a vector of policy choices $Y(\alpha, R_0)$ and a marginal cost of risk reduction $V(\alpha, R_0)$ that depend on acceptable risk $R_0$ and on the confidence level $\alpha$. In other words, cost-efficient policy choices are affected by adjustment for uncertainty. In particular, the marginal cost of risk reduction $V(\alpha, R_0)$ is a decreasing function of $\alpha$; thus, for any given level of marginal benefit, an increase in the confidence level implies a lower level of acceptable risk, i.e., a more stringent risk standard. The impact of an increase in the confidence level may be substantial: In a study involving contamination of drinking water wells in California by a pesticide, Lichtenberg, Zilberman and Bogen (1989) found that a one percentage point increase in the confidence level lowered the marginal cost of risk reduction by 25 to 35 percent.

One implication of this approach (and, indeed, any approach that takes uncertainty into account explicitly) is that optimal policy will consist of a portfolio of actions, some of which specialize in reducing adverse effects on average and others that specialize in reducing uncertainty. One example of the latter is research, ranging from basic research into the mechanisms leading to adverse effects in humans and the environment to data collection that would reduce estimation error. Lichtenberg and Zilberman (1988) show that uncertainty reduction is especially important when a higher confidence level is used, when there is less background uncertainty about damage and when toxicity is higher. Emphasis on reducing risk on average should be greater when the confidence level is lower, when there is more background uncertainty about damage and when toxicity is higher.

Lichtenberg and Zilberman's (1988) approach generates uncertainty-adjusted total and marginal cost curves for human health risk and/or environmental damage reduction. These cost curves can be used directly for risk-benefit trade-analysis. Alternatively, they can be combined with estimates of marginal benefits of risk/environmental damage reduction in a cost-benefit framework. Consistency suggests that marginal benefits, too, be adjusted for uncertainty, i.e., that upper confidence limits be used to generate benefits estimates.

One limitation of the Lichtenberg and Zilberman approach is that it treats cost as known with certainty. Yet we saw earlier that there is considerable uncertainty about pesticide productivity because it is subject to random influences and because measurement error can be substantial. Consistency once again suggests that these uncertainties be incorporated into a decision framework.

In Lichtenberg and Zilberman's framework, the level of confidence is treated as a fixed parameter predetermined by the regulatory agency. In reality, the choice of a confidence level (or margin of safety) is at bottom a policy decision. The agency must decide how reliable its decision will be, i.e., how great will be the chance that the risks are greater than estimated. Lichtenberg, Zilberman and Bogen's (1989) results indicate that this decision is not trivial: Increasing the confidence level increases the total cost and decreases the marginal cost associated with any risk standard substantially. Developing countries may be unwilling to bear the excess cost of meeting standards as reliably as is done in the U.S. Allowing less of a margin for error (choosing a lower confidence level) may provide a way of evaluating trade-offs in a more appropriate manner. Lichtenberg and Zilberman's (1988) theoretical results suggest that choosing a lower confidence level implies further that desirability of making usage regulation stricter and putting less emphasis on uncertainty reduction. For example, it may be efficient in developing countries to restrict access to pesticides to entities composed of professionals trained in pest control decision making and application methods.

*Acute versus Chronic versus Environmental Effects.* In the U.S., a principal lesson drawn from the environmental debacles associated with DDT and the organochlorines was that persistent compounds may have far-reaching, unforeseeable consequences. To minimize these ecological risks, USEPA has chosen to avoid persistent compounds in favor of pesticides that have relatively short lives. To be effective, these materials must be highly toxic, in contrast to the organochlorines, which have relatively low toxicity but are effective because of their long lives. Switching to short-lived, acutely toxic chemicals tends to increase the risk of acute human health effects, while reducing ecological damage and, conceivably long-term human health effects.

This posture may be best for the U.S. It is not clear that it is best for developing countries. The risk of poisonings from more acutely toxic materials should be higher in areas where education and skill levels are lower, e.g., in many developing countries. Anecdotal evidence suggests that storage and usage practices in these countries are highly unsafe. In addition, there is a tendency to use empty pesticide containers to store food and water. The U.S. places great stress on long term health effects such as cancer in regulating pesticides, but in countries where the life span is short, these long term health effects may not be a great concern, at least in the short run until income and longevity rise. These differences in posture may carry over into research and development of new pest management tools. Research and development by U.S. firms is shaped by EPA's regulatory stance; if the latter is inappropriate for developing countries, then the types of pest management tools being developed in the U.S. may be similarly inappropriate.

This is not to argue that developing countries are in fact best served by continuing to rely on organochlorines and other persistent pesticides. Damage to bird and fish populations from biomagnification of organochlorine residues may cause serious disruption to food production systems in developing countries. As noted earlier, there is some evidence that organochlorine residues can cause human health problems such as premature births, which may impose costs due to higher demand for medical care. However, those in developing countries, facing very different conditions in terms of income (and the marginal utility of income), life span and technical expertise may find it attractive to adopt very different approaches than the U.S. For example, it may make sense to restrict pesticide applications to a government-supervised entity of specially trained personnel rather than allow unrestricted access to acutely toxic materials.

**Toward a Broader View of Pesticide Policy**

In the U.S., at least, pesticide regulation is often thought of in terms of registration decisions and associated decisions about tolerances. Clearly, this view is far too narrow. Even in the U.S., pesticide use is influenced by research and development strategies, by agricultural policies, by governmental education and training initiatives and by the institutions used to deliver pesticides and pesticide use training. The use of integrated pest management (IPM) strategies in the U.S. has been made possible by a wide variety of public initiatives: land grant universities have been responsible for developing IPM strategies, for evaluating alternative strategies and for popularizing the "best" strategies in the farm community. Continued use of IPM in some areas depends on public provision of advisory services; the supply of pest control advisors is similarly dependent on the training capacity of land grant universities. Collective institutions of farmers like marketing boards have also taken responsibility for the development and promotion of IPM strategies; for example, the California Almond Marketing Board has been instrumental in promoting IPM methods among almond growers. More broadly, the context in which pesticide policy decisions are made is determined by public choices. In the U.S., for instance, farmers have been allowed unrestricted access to most pesticides, and pesticide manufacturers have been free to market pesticides to individual users. Laws like FIFRA specify the how USEPA can -- and cannot -- regulate pesticides.

Establishing a broad institutional framework for pesticide use decisions is thus a critical element in pesticide policy. Such a framework should include institutions for developing, evaluating and disseminating pest control practices best suited to a country; institutions for training farmers and pest control specialists in these new pest management technologies; oversight institutions to evaluate individual chemicals for efficacy and safety; and institutions to oversee disposal of empty containers and unused stocks. While the functions that such a framework needs to carry out may be the same in most countries, the specific form may not. In fact, alternative frameworks need to be evaluated in terms of the trade-offs they engender just like any specific policies. These institutional decisions may be more important than decisions about individual chemicals and the trade-offs involved should thus be scrutinized more closely.

One essential component for making evaluations of trade-offs for pesticide policy is a capacity for interdisciplinary policy modeling. Decisions about trade-offs must be based on the best available information on the consequences of alternative policy choices, which is necessarily derived from a variety of disciplines in the biomedical, crop and decision sciences. This information must be melded into models that can be used for policy analysis. Conscious cooperation is needed because different disciplines take different, often incompatible, approaches to modeling. For example, we argued that uncertainty is a central feature of most elements involved in pesticide policy, so that trade-off evaluation need to incorporate uncertainty explicitly. The Lichtenberg and Zilberman (1988) approach has the attraction of using concepts of uncertainty that are amenable to the natural sciences. But implementing this approach requires having risk estimation models that include parameters measuring the effects of policy interventions on the mean and standard deviation of risk. This type of modeling requires active cooperation between economists and public health experts. Cooperation between economists and crop science experts (agronomists, horticulturists, entomologists, plant pathologists) is equally necessary. Estimating the benefits of pesticide use requires models of the essential components of crop ecosystems that include the impacts of alternative policy choices on yields, models predicting changes in farmers' reactions to different policy alternatives and models of the markets affected by changes in pest management technology.

The need for conscious development of this sort of interdisciplinary modeling capacity is illustrated by Lichtenberg, Spear and Zilberman's (1991) attempt to model the costs and benefits of re-entry regulation. To protect farmers and field workers from toxic residues, USEPA forbids entry into treated fields until residues have decayed to acceptably low levels. The period of time during which entry is forbidden is called a re-entry interval (or, at the end of the season, a pre-harvest interval). Analyzing the trade-offs involved in re-entry regulation requires knowledge of (1) crop growth and the productivity of field operations during the season, (2) decay rates, exposure and toxicity of pesticide residues and (3) how re-entry regulation affects farmers' pesticide use and operating decisions. Re-entry regulation affects principally fruit and vegetable crops. There are very few models of fruit and vegetable crop growth. With few choices available, Lichtenberg, Zilberman and Spear settled on apples, for which a highly simplified model was available. Data on residue decay, worker exposure and the probability of acute poisoning were available for the insecticide ethyl parathion, which has been associated with numerous poisoning incidents. The study focused on insecticide use to control end-of-season infestation of codling moth. Data on prices were used to estimate price trends over the growing season. These models were combined to estimate trade-offs between numbers of poisonings suffered by harvest workers and revenue losses suffered by growers as a function of the length of the pre-harvest interval. Information on avoided medical cost was used to obtain a conservative estimate of an optimal pre-harvest interval.

This effort produced some interesting qualitative insights. The behavioral model of pesticide application decisions showed that re-entry regulation introduces scheduling rigidities that could induce farmers to rely on preventive treatment. The risk assessment model indicated that rainfall reduced residue levels, suggesting that one could reduce the cost of achieving any given level of farmworker protection by making the length of the pre-harvest interval dependent on the amount of rain occurring after the time of pesticide application. Unfortunately, parathion is used on less than a third of U.S. apple acreage and is not used at all in Washington and California, two major apple growing states. Lack of surveillance data on farmworker poisonings ruled out empirical validation of the risk assessment model. Insufficient information on crop growth and pest dynamics restricted the analysis to very simple pesticide use decisions. Thus, the model they developed is useful mainly to illustrate methodology.

The lesson to be drawn from this example is that interdisciplinary modeling can yield some important insights for policy makers. To be fruitful, interdisciplinary work must focus on answering policy-related questions. The models used need to capture the principal elements governing pesticide productivity, economic decision making and adverse health and environmental effects -- but no more. The point is to help guide decisions, not replicate faithfully any of these processes. Collaboration between scientists of different disciplines can, by itself, generate new insights. The process of building decision models of this kind will also highlight data needs and help prioritize data collection. In sum, pesticide policy needs reliable, comprehensive forms of analysis. The trade-offs involved are great enough that information -- about production, about behavior, about markets and about health and environmental effects -- can make a significant difference.

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# Figure 1
## Market Impacts of Pesticide Regulation