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ECONOMICS OF PESTICIDES, SUSTAINABLE FOOD PRODUCTION, AND ORGANIC FOOD MARKETS

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1. ADOPTION AND DIFFUSION OF SUSTAINABLE FOOD TECHNOLOGY AND POLICY

Darwin C. Hall and L. Joe Moffitt

I. BEGINNINGS OF SUSTAINABLE FOOD TECHNOLOGY AND POLICY – AN EARLY HISTORY

In the late 1800s, a philosophical battle raged between advocates of chemical controls for pest management and advocates of biological and cultural controls. "Charles W. Woodworth, Professor of Entomology at the University of California, advocated an ecologically based pest management approach throughout his long career. In 1896, he stated that everyone should have a clear idea of the controls available and how to apply them" (Smith, 1978).

A. Integrated Control and Integrated Pest Management

Then, following the Second World War, agricultural technology underwent a revolution with increasing applications of water, chemical fertilizers, pesticides and mechanization. Agronomic research focused on crop varieties best suited to exploit low-priced inputs, with dramatic increases in yield per acre. In the United States, the institutions propelling the technological revolution were financed by the national government, subsidizing research at land-grant universities in each state. Since the passage of the Hatch Act in the 1800s, every college of agriculture has faculty with joint appointments in agricultural experiment stations to carry out basic research, and agricultural extension

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positions to deliver the results to growers and assist with adoption of new technology. Major chemical companies and farm equipment manufacturers donated increasing amounts of funding, helping to shape the kinds of technological improvements borne of this institutional system. Headley (1968) found that increases in yields more than offset the costs of pesticides to growers. As he predicted, the production and sales of pesticides doubled in the next ten years just as it had done in the prior decade (USDA 1964, USDA 1971–1977).

In the 1950s, a group of entomologists in the United States changed the course of technological innovation. They had concerns about adverse impacts of agricultural chemicals on workers, consumers, fish and wildlife, and to the agricultural ecosystem itself. Knowing that economics drove growers' decisions, these entomologists focused on adverse effects to the agricultural ecosystem, and consequent economic loss, and they searched for pest controls that would be economic alternatives to pesticide applications (van den Bosch, Reynolds & Dietrick, 1953; van den Bosch & Dietrick, 1953, 1957; van den Bosch, Schlinger & Dietrick, 1957, 1959; van den Bosch, Schlinger, Dietrick & Hall, 1957; van den Bosch, Schlinger, Dietrick, Hagen & Holloway, 1959). By the mid-1950s, some graduate students left their Ph.D. programs in entomology and opened pest management consulting practices and insectaries (e.g. Dietrick), selling advice and biological pest controls to growers. As early as 1954, the entomologists revolutionizing agricultural research had coined a term to describe their philosophy, "integrated control" (Smith, 1978). Bottrell (1979) credits Bartlett (1956) as the first to publish the term, integrated control.

In their seminal paper, Stern, Smith, van den Bosch, and Hagen (1959) defined the concept of integrated control as "pest control which combines and integrates biological and chemical control" (p. 86), where pesticide application is "based on conclusions reached from periodically measured population densities of pests and beneficial species, . . . and based on a sound knowledge of the ecology of the organisms involved and projected future population trends of pests and natural enemies" (p. 87). The concept of integrated control was subsequently broadened to include all control methods (Smith, Apple & Bottrell, 1976) and formally renamed Integrated Pest Management (IPM) by the Council on Environmental Quality (1972) of the President. From an economist's perspective, "IPM substitutes knowledge and information for pesticides by optimally choosing from a wider set of available actions; considering interactions between pests, natural enemies, weather patterns and crop growth and utilizing more accurate knowledge of such interactions;

monitoring insect and mite populations in a timely and precise fashion; and utilizing more accurate monitoring methods and devices" (Hall, 1977a).

The role of IPM can be juxtaposed to the "pesticide treadmill", caused by resurgence, secondary out-breaks, and resistance. Pesticide applications kill and reduce populations of pests and natural enemies (parasites, predators), and then the pest populations resurge to much higher levels, because there is a time lag until food sources are available for predator and parasite populations to resume previous levels. Unable to wait for natural control to reestablish itself, the farmer is compelled to apply more pesticide. Some potential (secondary) pests are controlled by natural enemies to populations below levels that cause economic damage. Pesticide application kills the natural enemies of secondary pests, and their populations resurge to levels that cause economic damage, compelling even more pesticide applications. Arthropods are small, and the smaller the creature, the shorter the life span, and the more fecund, with greater "genetic plasticity". With pesticide application the survivors that reproduce have some inherent resistance, and they reproduce by the thousands within short time periods. Repeated exposure results in pest resistance to the pesticide, compelling higher dosage rates (Carlson, 1977). Ever more frequent applications at higher doses define the pesticide treadmill. The alternative, IPM, considers the economic costs of resurgence and secondary outbreaks when making the economic decision to use chemicals, and requires monitoring the agro-ecosystem to measure the level of pest infestations and to identify populations of beneficial insects – augmenting them with releases of biological controls and/or maintenance of habitat or crop rotation conducive to enhancing the populations of beneficial insects, timing pesticide applications to destroy pests when they are most vulnerable or avoiding applications when biological controls might be vulnerable, and applying pathogens to kill pests, for examples. The reduction in pesticide use slows the rate of pest resistance.

Research entomologists claimed that advances in IPM strategies could increase yield and reduce pesticide applications and costs (see above references to Van den Bosch et al.). From 1972 to 1978, the National Science Foundation, the Environmental Protection Agency, and the U.S. Department of Agriculture funded a $5.5 million, 18 university research project lead by Carl B. Huffaker (Huffaker, 1978). In the late 1970s and early 1980s, Perry L. Adkisson lead the continuation of the research in IPM; the Adkisson project became known subsequently as the Consortium for Integrated Pest Management, reporting successful IPM programs for cotton, alfalfa, soybeans, grapes, and apples (Frisbie & Adkisson, 1985). In Europe, research on IPM and low input alternatives has helped growers substitute alternatives for pesticides (OECD, 1993, 1994a).

Fungicide use remained relatively stable during the 1980s, and insecticide use actually dropped as a result of the development and adoption of integrated pest management (Carlson, 1988; Zilberman, Schmitz, Casterline, Lichtenberg & Siebert, 1991).

Concern about the widespread use of pesticides persists for at least two reasons. Pesticides are unique among intentionally introduced environmental contaminants in that they are specifically designed to be injurious to living organisms. Agricultural uses of pesticides can involve direct risks of residues in food and water potentially ingested by humans. As an example, the safety of apples that contain chemical residues for consumption by children has produced a vigorous debate in the United States and has raised further concerns about the appropriate management of pesticides in the environment (Natural Resources Defense Council, 1989). There is also concern about potential adverse impacts on wildlife and environmental resources due to pesticides. Moreover, water quality has been a continuing pesticide-related concern among environmentalists. The environmental and human health concerns have contributed to interest in economically efficient pesticide use among farmers, researchers, environmentalists, the general public, and economists as well.

Because of the concerns about pesticides in the environment, it is perhaps not surprising that concurrent with the definition of integrated pest management was the development of interest in agricultural pest management among economists. The first economic analysis of pest control in agriculture (Hillebrandt, 1960a, b) appeared one year after the pathbreaking definition of integrated control by the entomologists, Stern et al. (1959). Patricia Hillebrandt's work is not only noteworthy for being the first economic analysis of its kind but also for foreshadowing the important role that female economists would come to play in pest management economics research. In the decades that followed Hillebrandt's seminal piece, important contributions by economists such as Christine Shoemaker (1973a, b), Katherine Reichelderfer-Smith (1979), and Carolyn Harper (1989, 1992) would prove to be influential in the field.

IPM challenged the eradication philosophy inherent in chemical control with the concept of pest control. Instead of eradication, the idea was to find the "economic threshold", originally defined by economists (Headley, 1971) as the level to which the pest population is reduced by controls, although the common meaning of "threshold" used by entomologists is the population level at which pesticide applications are initiated (Hall & Norgaard, 1973, 1974). The first econometric application, Carlson (1970) found that adjusting applications to a forecast of the pest population, rather than calendar spraying, reduces pesticide use and increases expected profits. Casey, Lacewell and Sterling (1975) found that profits increases for farmers who reduced pesticide use when beneficial insects are present.

IPM as a strategy means to control pests with a combination of controls: biological, mechanical, cultural, chemical, genetic and legal. However, the early work on the economics of pesticides tended to focus on basic functional relationships involved in crop protection from pests and rarely ventured into the realm of deployment of an arsenal of controls deployed by pest management consultants, as envisioned by integrated control's entomological founders, with some exceptions (e.g. Willey, 1974; Hall, Norgaard & True, 1975; Hall, 1977a, b, 1978; Hall & Duncan, 1984; Carlson, 1980). Moreover, IPM became more generally accepted by conventional agriculture, and by the end of the 1980s, Zilberman et al. (1991) estimate that "more than 50% of California growers practice IPM in one form or another." The practice of IPM, however, did not easily integrate biological and chemical controls; to the contrary, Carlson (1988a) found substantial obstacles to the adoption by growers of biological controls. IPM decision strategies can be defined solely in terms of pest decision making that maximizes profit to growers, or more broadly by taking into account negative external costs to consumers (food safety), worker poisoning, contamination of drinking water, contamination of fish, and more general environmental damage via transport by air, surface and ground water. As Moffitt (1993) states, "One can apply IPM threshold decision making principles to define a rational pest control strategy that growers might accept but that environmentalists might not." Interest turned toward organic farming (Carlson, 1988b).

B. Organic Farming

With the exceptions of some religious groups and the 1960–1970 counter-culture, few believed that organic farming was economically viable. Organic farming pioneers began the search for farming methods that do not rely on synthetic chemicals. In 1970, Rodale Press in Pennsylvania began a program to certify organic food, and in 1971 published a list of 34 organic farms in California. Some of these growers established the California Organic Growers, and in 1973, 50 growers reorganized this organization into the California Certified Organic Growers (CCOF). By 1988, CCOF had 380 growers and continued to expand rapidly. Cook (1988) estimates that in 1987 there were approximately 900 growers of organic products in California, equal to about 1% of the industry total of 82,463 (1982 Census of Agriculture), with revenue to growers of $50 million from 30,000 acres in California (relative to 7,831,307 total acres farmed). This represents very significant growth relative to 1982 when Altieri et al. (1983) estimated that there were 273 growers of organic products. "CCOF currently has about 736 growers that are CERTIFIED currently, and about 849 members certified if you include our processors. These numbers do not include pending or transitional members of CCOF" (Brian Sharpe, CCOF, September 2001).

With little support beyond their own organizations, growers have succeeded in devising complex strategies to grow and market organic food, and survived for more than a decade. Their survival is noteworthy given the comparable institutional superstructure that supports the research, development, registration, certification and application of synthetic chemicals. Partially in response to the needs of organic growers, the University of California established the Sustainable Agriculture Research and Education Program (SAREP) in 1986. Other states have developed similar programs, and in fiscal year 1988 a national program began, called Low-Input Sustainable Agriculture (LISA) and now referred to as Sustainable Agriculture Research and Education (SARE).

II. POLICY TO REDUCE NEGATIVE EXTERNALITIES FROM AGRICULTURAL CHEMICALS

The economics literature on policy relevant to integrated pest management is slim. The literature prior to 1981 is described in Osteen, Bradley, and Moffitt (1981) while a description of some key research/policy studies prior to 1993 is contained in Moffitt (1993). More policy-related material can be found in the early literature on policy related to organic farming (see Hall, Baker, Franco & Jolly, 1989, and the references therein).

Baker (1987, 1988, 1989) emphasized that price support policies based on acreage or yields encourage pesticide use, an issue picked up by Shortle and Abler (1999). Shortle and Abler (1999) mention non-point source run-off of pesticides to ground and surface water, contaminating groundwater in North America and Europe (OECD, 1991), and causing increased costs for water treatment and adverse impacts on human health. They also note damage to fisheries and ecosystems. Opschoor and Pearce (1991) point out that pesticides are persistent in the environment, with long half-lives; absorbed by microscopic animals and plants at the base of the food chain, they biomagnify as they pass through the food chain, stored in fatty tissue; and they are ubiquitous, found everywhere in the environment from close to the point of application to the polar regions of the earth. Thought to travel through water and the food chain, more recent work shows that pesticides travel on air currents across and between continents (AMAP, 1998; Raloff, 1996).

Shortle and Abler (1999) list the standard policy options: emission standards or taxes, and tradable discharge permits, but the costs of monitoring are prohibitive. They also suggest voluntary adoption of environmentally favorable technology, "combining public persuasion with technical assistance," but this approach is not economic, limiting success. They also mention pesticide registration, cancellation, and labeling so as to restrict use to safe practices or to avoid applications that risk human health or environmentally sensitive areas. Finally, they mention the option of taxes on pesticides, but note taxes on agricultural inputs have been so low as to have almost no impact, except in Sweden (OECD, 1994b) and Iowa in the United States.

Zilberman et al. (1991) argue that the risk to food safety is best addressed by labeling laws that distinguish among foods grown by low input methods and those that are organic. They list these policies to mitigate risk to workers, water contamination, and the environment: "chemical bans, use restrictions, pesticide fees or taxes, subsidies for non-chemical pest management practices, protective clothing, and application standards." They argue that uniform standards (bans, or uniform standards across crops and regions, such as use restrictions, protective clothing, application standards) are inferior to pesticide fees when trading off between policy costs and risks to human health and the environment.

Hall et al. (1989) consider the widest array of policy options: effluent charges, tradable permits, input taxes, subsidies, torts, food labels, application safety restrictions, and selectively banning or restricting use of pesticides. They review the options of effluent charges or selectively banning pesticides or restricting use, but dispersed use makes measurement and tax collection or compliance difficult to enforce, and the optimal charges or restrictions require information too costly for government to obtain. They note that food labels fail to include complete and accurate information of pesticide and byproduct residues, and experts disagree over risks. For farm workers, safety restrictions require literacy of workers and active participation by the growers, a principle-agent problem; torts will not solve the problem for illegal workers, direct and indirect exposure and cause and effect makes the burden of proof a separate problem (Tietenberg, 1988). More generally, Menell (1991) states that torts result in "highly unsystematic levels of compensation, distorted incentives, and high transactions costs." While Hall et al. argue that pesticide taxes or subsidies for low input or organic farming are the best options, they note that pesticide taxes redistribute wealth from chemical companies and growers, and make growers less competitive relative to other countries, reducing the political feasibility of this option.

Hall et al. (1989) also consider three policy options to increase the efficiency of organic food markets. One option is to subsidize research on marketing opportunities for supermarkets to sell organic food. A second option is to expand the existing Federal State Market News Information Service to provide data on prices, available distributors, wholesalers, and producers of organic food, not just conventionally grown food. A third option is to legally define several alternative labels, not just organic, but also CCOF (California Certified Organic Farmer) certified, chemical free, and IPM food.

One final area that the literature simply ignores is policy proposals to encourage the adoption of IPM.

III. IPM RELATED POLICY – AN EARLY HISTORY AND SOME MODEST PROPOSALS

One of the entomologists who invented the concept of integrated control, Robert van den Bosch argued in favor of prohibiting licensed pest control advisers from having a financial interest in the sale of pesticides. His position was that the conflict of interest inherent in a medical doctor not acting as a pharmacist is an apt analogy to the case of pesticides (van den Bosch, 1978). Hall (1977a, b), Burrows (1983), and Hall and Duncan (1984) estimate that growers who rely on advice from independent pest management consultants use about 50% less pesticide, have a slight reduction in yield, and are as profitable as growers who rely on pesticide salesmen for advice.

Hall (1977a) refined van den Bosch's idea. Instead of prohibiting pest control advisers from selling pesticides, create two classes of licensed advisers, those who are chemical salesmen and those who have no financial interest in pesticide sales. Hall pointed out that, in California, recommendations from licensed advisers must be in writing, and the data are tabulated. The data could, therefore, be summarized by crop and licensed adviser and then provided to growers to aid in their choice of an advisor. In 1980–1981, the Director of the California Department of Food and Agriculture worked with the California Legislature to craft a bill to implement this policy, but the bill was defeated. Had this bill passed, it would have been possible to alter restrictions on the uses of pesticides, allowing in certain cases the use of pesticides with a prescription from an independent pest control adviser.

In another refinement to van den Bosch's idea, Hall (1977a) suggested that EPA could allow the use of severely restricted pesticides in special cases where the pesticide is part of an IPM program and its use demonstrably and substantially reduces the total amount of pesticide applied. The EPA Administrator permitted, for a time, the use of a chlorinated hydrocarbon to control ants in citrus. Not a pest, ants eat a predacious mite, interrupting the biological control of a mite pest, and triggering the pesticide treadmill. Moffitt (1993) recounts another application of this idea in Massachusetts. There, 24 active ingredients are restricted from being applied in areas that contribute water to a well under severe recharge and pumping conditions or within a one-half mile radius of public drinking water wells that supply more than 100,000 gallons of water per day. A variance permits use of banned pesticides under certain conditions, including IPM pest monitoring and selective pesticide use.

Economic assessment of sustainable agricultural practices has continued to mature and to add to our understanding of how we might design policies to ensure an adequate and diverse food supply. This volume presents some of the recent developments and applications in this field.

IV. CONTRIBUTIONS IN THIS VOLUME

This volume is divided into four sections focusing respectively on pesticide use, sustainable food supply, demand for sustainable food production, and related policy. These four sections encompass the range of advances in theoretical and applied economic analyses concerned with pesticides and sustainable food markets. Chapter contributions include different methodological, ideological, and geographical perspectives.

A. Pesticide Use

The section on pesticide use contains four chapters that reflect recent trends in economic modeling related to pesticide use. Preceding even the appearance of Rachel Carson's (1962) influential Silent Spring, this area of economic research has its roots in the very beginnings of what is referred to currently as the economics of environmental resources. In the first chapter in this section, Hall and Moffitt reconsider a traditional topic in the economics of pest control; viz., the econometric measurement of the marginal product of pesticide. This econometric problem dates back to some of the earliest studies (Hillebrandt, 1960a, b; Headley, 1968) in the economics of pest control. Their reconsideration challenges what has become an accepted notion since the mid-1980s that the functional form describing production leads to an unambiguous directional bias in the econometric estimation of the marginal product of pesticide. Their analysis shows that, contrary to current perceptions in the economics of pesticides literature, sweeping econometric generalizations concerning the superiority of popular production function forms is not currently possible. They clarify misconceptions concerning the functional form issue and extend related econometric methods to account for critical biological features including pest numbers and phytotoxic effects of pesticides on crops. An empirical example illustrates their extensions to popular econometric practices.

In the second chapter in this section, Fernandez-Cornejo and Pho focus on the role of economic incentives in pesticide use, which has also been a traditional theme in the economics of pesticide use. Utilizing time series observations from 1945–1994, they provide the first direct econometric test of the induced innovation hypothesis as an explanation of the rapid increase in the use of commercial, chemical herbicides since the Second World War. The hypothesis tested is that relative prices are determinants of technical change and factor bias. A unique aspect of their econometric analysis is that it is based on quality-adjusted price and quantity data for both herbicide and labor variables. Their elasticity estimates tend to agree with the induced innovation hypothesis with respect to labor and land; however, the same cannot be said for herbicide/machinery substitution. Moreover, apparent inadequacies in their data series on private research expenditures also result in findings contrary to expectations. Their extensive data development and modeling effort highlights the difficulty in explaining the economic rationale behind one of the most obvious trends in pesticide use.

The third chapter by Davis and Tisdell looks at the status of farm-level decision strategies intended to promote efficient use of pesticides. They survey the economic threshold concept in agricultural pest management, including development of the concept in early works and recent extensions to account for multiple pest species and pest resistance to pesticides. Of special interest is their diagrammatic rendering of some steps toward producer optimal decision making in a multiple pest context. They appraise the potential for applying more sophisticated management methods versus routine efforts to reduce pesticide use in the livestock industry.

Wiebers, Metcalfe, and Zilberman continue the focus on understanding pesticide use by quantifying the expected difference in insecticide use per acre between growers who rely on pesticide salesmen as pest control advisors versus grower treatment recommendations in California tomato production. Their conceptual framework posits relationships between different pest control variables leading to a limited dependent variable econometric model. The model is estimated using data from a survey of tomato growers in six northern California counties. Results of estimation suggest that insecticide treatment recommendations from pest control advisors involve more insecticide use than growers' own treatment decisions. Based on the empirical analysis, some novel suggestions to reduce pesticide use include separating the sale of pesticides from pest control advice and raising pest control advisors' perceptions of grower expertise by improving grower training related to pest control.

B. Sustainable Food Supply

The section on sustainable food supply contains two chapters from both European and North American perspectives. In the first chapter in this section, Michelsen provides a current economic perspective on organic farming in Europe and, in particular, addresses the variation in responses to the Common Organic Farming Policies introduced by the EU member countries in 1992. The variation in the development of organic farming in member countries is significant in terms of farm size, production, and regional distribution, although the focus of Michelsen's analysis is on the differential impact of policy on the number of organic farms in member countries. The policies Mitchelsen examines include a common EU-wide definition of organic farming concomitant with certification and the obligation of member countries to provide some level of financial support for organic farmers. Michelsen explains the variation with institutional and organizational theory. He provides empirical analysis of the impact of various policy instruments on the growth of organic farming, the presence of institutional conditions for policy change, and comparison of institutional interrelationships with regard to organic farming policy.

In the second chapter in this section, Klonsky and Smith provide a unique analysis of the economics underlying the rapidly growing organic food industry in California, a state that accounts for more than half of organic vegetable production in the United States. They observe that aggregate growth figures tend to mask the significant changes in the composition of the organic farming sector that occur due to the substantial entry of new and exit of existing organic farms. Using detailed data from California's organic farming registration program gathered for the period 1992–1997, they use a random utility econometric model to compare farms that entered and exited the organic sector during this period. An important finding of their analysis is that entering organic farms are likely to be small compared to existing members of the organic industry. They discuss implications of the recently enacted federal organic certification program for entry and successful marketing by new organic producers and for the private certification industry.

C. Demand for Sustainable Food Production

The section on demand for sustainable food production provides a clear transition to policy issues. Padel, Lampkin, Dabbert, and Foster provide a comprehensive review of organic farming policies in the European Union with the aim of assessing whether and how organic farming can contribute to existing policy objectives. In their approach to policy evaluation, they take as the foundation for policy analysis the actual and proclaimed objectives of politicians, such as maintaining farm income, minimizing environmental consequences of farming, and enhancing rural development. Existing empirical evidence sheds light on factors driving sustainable food production, especially political demand for a cleaner environment.

Vanzetti and Wynen focus on the demand for domestic versus imported food products. They question whether environmental concerns should always contribute to the demand for domestic food products. Vanzetti and Wynen make the case that, contrary to some popular contentions, trade can contribute to a more environmentally sound way of supplying agricultural products to consumers. They illustrate their case with an example from the international wheat trade.

D. Policy

The last section of this volume focuses on policy related to pesticides and sustainable food production. In the first chapter in this section, Lynch and Carpenter present an economic analysis of regulated use in California of a chemical replacement for the fumigant methyl bromide. They compare three different policies (quotas, first come – first serve, and highest-value) for restricting use of 1,3-D at the township level based upon the criteria of efficiency and distributional (crop and county) impacts, using a putty-clay production framework and a model of constrained grower decision-making. An especially interesting finding of the analysis is that a quota based on historical use maximizes the aggregate value of 1,3-D in several different California counties.

The final two chapters in this section provide an interesting contrast in methodologies for economic evaluation of pesticide policies that account for human health considerations explicitly. Wilson, in the second chapter in this section, uses contingent valuation to measure the costs of farmer exposure to pesticide in Sri Lanka and finds the cost to be significant. He shows that willingness to pay increases with adverse farmer experience in handling pesticides, concluding that government intervention to educate growers about the hazards of pesticide may be valuable public health policy.

In the final chapter, Sunding and Zivin pursue a different approach to measuring the cost of pesticide exposure. They combine economic and toxicological information in an analytical framework for comparing the economic efficiency of pesticide regulatory policies. Explicitly incorporating contamination, exposure, and dose response factors in their analysis suggests that policies that affect both contamination and exposure are more efficient than policies that affect only one factor.

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2. MODELING FOR PESTICIDE PRODUCTIVITY MEASUREMENT*

Darwin C. Hall and L. Joe Moffitt

ABSTRACT

This paper revisits claims about implications of production function specification for pesticide productivity measurement and presents two extensions of the popular damage control specification, along with an empirical application. One extension eliminates bias caused by relying solely on economic data and shows how to include variables that represent the pest population when they are necessary to avoid bias. The second extension allows for the possibility of phytotoxicity. These extensions generalize the damage control specification by eliminating bias and allowing for a range over which the first and third stages of production may occur. The main contributions of the analysis are to clarify existing misconceptions about pesticide productivity modeling and to provide extensions to the damage control specification that permit greater realism for empirical analysis.

* An earlier version of this paper was presented at a pesticide economics conference held at the University of Massachusetts in honor of the memory of Carolyn R. Harper.

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1. INTRODUCTION

Virtually all types of management strategies related to pest control in agriculture require knowledge of the impact of pest density on crop production and the relationship between density and pest control inputs (e.g. Marsh, Huffaker & Long, 2000; Saphores, 2000; Sunding & Zivin, 2000). The estimated form of such relationships can be critical for farm-level decision making and for public policy analyses as well. Not surprisingly, the econometric specification of such models has been the subject of a number of studies (Lichtenberg & Zilberman, 1986; Carrasco-Tauber & Moffitt, 1992; Du et al., 2000).

This paper begins by clarifying misconceptions that have resulted from conceptual problems underlying the damage control model formulated by Lichtenberg and Zilberman (1986). Following this, we pursue important extensions to the damage control model for empirical work. The model can be extended in one way by explicitly recognizing the process by which inputs control for damage. The extension in this paper reveals a second source of bias when estimating the pesticide production/cost relationship solely with measures of outputs, inputs, costs and/or prices. If the inputs to control damage are not applied prophylactically, then the estimators proposed by Lichtenberg and Zilberman are biased; to avoid bias it is necessary to specify the damage and control relationship and measure the state variables (pest population) that cause the damage. Economic variables alone are insufficient; to estimate pesticide productivity it is necessary to measure and model variables from Mother Nature, as well as economic variables.

A second extension of the damage control specification is essential to estimate the productivity of herbicides. If the herbicides have phytotoxic effects, then the estimators proposed by Lichtenberg and Zilberman are biased; this second extension avoids bias caused by phytotoxicity. More generally, the two extensions in this chapter account for circumstances where the input that controls damage may also cause damage.

2. DAMAGE CONTROL MODEL AND PRODUCTIVITY

In their frequently cited study, Lichtenberg and Zilberman (1986) argued that pesticide inputs should be entered into econometric production function models in a different manner than other inputs. Their suggestion was to encapsulate the pesticide variable in an abatement function; i.e. a function that maps the pesticide variable onto the unit interval, and to enter the abatement function rather than the pesticide variable into the production function. We refer to their model as the damage control model. Some earlier studies (e.g. Moffitt & Farnsworth, 1981) used the same kind of functional specification; however, Lichtenberg and Zilberman (1986) were the first to attempt to provide an econometric rationale for using such an abatement relationship. A number of pesticide productivity studies done subsequent to Lichtenberg and Zilberman (1986) have also used their notion of a damage control model.

As measured by its apparent impact on subsequent literature (e.g. Fox & Weersink, 1995), perhaps the main substantive conclusion of Lichtenberg and Zilberman (1986) is that the use of a standard Cobb-Douglas type production function to estimate pesticide productivity leads to overestimation of the marginal product of pesticide. More specifically, they conclude that if the damage control model is the true model, then least squares estimates of the marginal product of pesticide derived from a Cobb-Douglas production function will overestimate the true marginal product of pesticide for all values of pesticide above the geometric mean of pesticide use contained in the statistical database. Figure 1 depicts the main result diagrammatically. While several criticisms have been leveled at various features of the Lichtenberg and Zilberman (1986) analysis (e.g. Pandey, 1989; Blackwell & Pagoulatos, 1992), the main conclusion depicted in Fig. 1 has apparently never been disputed.

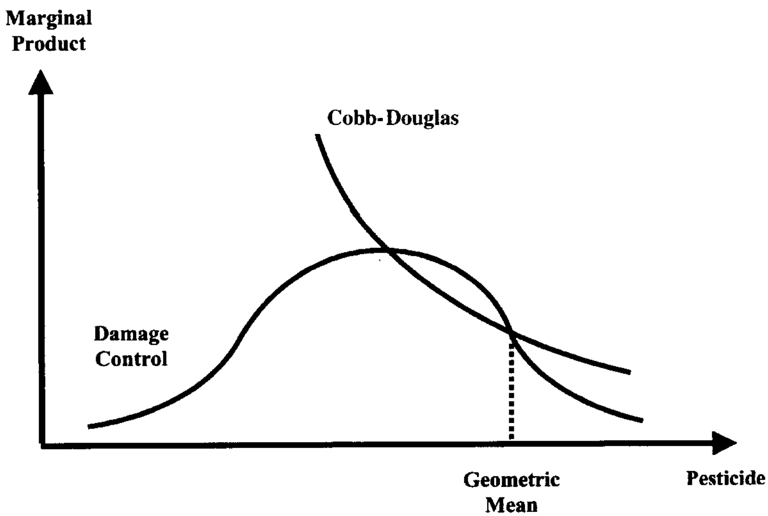

Fig. 1. Alleged Relationship Between Marginal Products Estimated using Cobb-Douglas and Damage Control Models.

As shown below, because of a conceptual problem in the Lichtenberg and Zilberman (1986) derivations, their analysis does not support the main conclusion of their study (Fig. 1) that is, in fact, incorrect. This observation is perhaps not surprising when one abstracts from the context in which their study was conducted and views their model simply as a regression relationship. So, before getting into details, note that in the abstract their study purports to show that the slope of a nonlinear functional form will be overestimated if least squares regression and a log-linear form are applied erroneously to the data generated by the true nonlinear form. When viewed from this perspective, it is apparent that this is a very tall order. While the particular nonlinear functional forms for which the Lichtengerg and Zilberman (1986) result holds are unclear, it is clear from the following analysis that the result does not hold when the nonlinear functional form is the damage control model.

The main criticism of the Lichtenberg and Zilberman (1986) conclusions about relative magnitudes of marginal products estimated using the Cobb-Douglas and damage control models is based on the fact that they draw their conclusions from comparisons with numbers that are not estimated marginal products. The main flaw in the Lichtenberg and Zilberman (1986) reasoning occurs in their Eqs (A7) and (A8) (Lichtenberg & Zilberman, 1986, p. 273). Some discussion of marginal product and its evaluation is useful to see the conceptual problem embodied in their reasoning. The marginal product (MP) of an input can, of course, be evaluated at any level of input use. Lichtenberg and Zilberman (1986) defined Q as output, Z as a vector of ordinary inputs, and X as a damage control input. Using the Lichtenberg and Zilberman (1986) notation, if the production model is $Q = F(Z, X) = \alpha Z^{\beta} X^{\gamma}$, then $MP_X = \partial Q/\partial X = \gamma\alpha Z^{\beta} X^{\gamma-1} = \gamma F(Z,X)/X$ which obviously depends on the values of Z and X. In particular, MP_X at the arithmetic means $\bar{Z}$ and $\bar{X}$ of the input variables, say $MP_X(\bar{Z}, \bar{X})$, is $\gamma F(\bar{Z}, \bar{X})/\bar{X}$ while MP_x at the geometric means, Z^* and X^*, say $MP_X(Z^*, X^*)$, is $\gamma F(Z^*, X^*)/X^*$. Note that the arithmetic mean of a variable X given n observations is defined as $(1/n)\Sigma_{i=1}^{n} X_i$ while the geometric mean is defined as $\Pi_{i=1}^{n} X_i^{1/n}$.

Marginal products for inputs in production models estimated by ordinary least squares (OLS) in double-log form are commonly reported at the geometric means. This point is selected for evaluation because, due to a property of OLS, the marginal product calculation is simplified at the geometric means. To see this, note that in the Lichtenberg and Zilberman (1986) notation, the double-log form of $Q = \alpha Z^{\beta} X^{\gamma}$ is $\ln Q = a + \beta \ln Z + \gamma \ln X$, where $a = \ln \alpha$. Given n observations on Q, Z, and X, OLS parameter estimates are known to provide an exact fit at the arithmetic means of the data variables (here data variables are logarithms – $\ln Q$, $\ln Z$, and $\ln X$); that is,

$(1/n)\Sigma_{i=1}^{n} \ln Q_i = \hat{\alpha}_{OLS} + \hat{\beta}_{OLS}(1/n)\Sigma_{i=1}^{n} \ln Z_i + \hat{\gamma}_{OLS}(1/n)\Sigma_{i=1}^{n} \ln X_i$. Exponentiation of both sides gives $Q^* = \hat{\alpha}_{OLS}Z^{*\hat{\beta}_{OLS}}X^{*\hat{\gamma}_{OLS}} = F(Z^*, X^*)$; that is, the geometric means of output and the inputs are a point on the fitted production function. Since $Q^* = F(Z^*, X^*)$, marginal product at the geometric means, $\hat{\gamma}F(Z^*, X^*)/X^*$, can be evaluated as $\hat{\gamma}Q^*/X^*$. This simplification perhaps explains why the geometric means have traditionally been used for reporting estimated marginal products for Cobb-Douglas type production functions fitted in double-log form by OLS (see e.g. Headley, 1968).

The very convenient calculation of MP_X at the geometric means is facilitated by a substitution based on the fact that $Q^* = F(Z^*, X^*)$. However, this type of substitution is not possible at any other level of input use for the OLS estimated double-log case. In particular, use of the arithmetic means of output and the inputs to form $\hat{\gamma}\bar{Q}/\bar{X}$ as MP_X is inappropriate because $\bar{Q} \neq F(\bar{Z}, \bar{X})$ in finite samples or asymptotically. It is important to note that $\hat{\gamma}\bar{Q}/\bar{X}$ and related expressions contained in Eqs (A7) and (A8) of Lichtenberg and Zilberman (1986) are not marginal products by definition. Hence, their relative magnitude does not predict the relative magnitude of marginal products that will be forthcoming from use of Cobb-Douglas type and damage control specifications of production. It is use of this erroneous substitution by Lichtenberg and Zilberman (1986) in their Eqs (A7) and (A8) that leads to the erroneous conclusions drawn concerning the relative magnitudes of marginal products estimated using Cobb-Douglas type and damage control production models.

Empirical confirmation of the above is provided by results contained in Carrasco-Tauber and Moffitt (1992, p. 160). Table 1 shows their estimated marginal products for various farm inputs using both a Cobb-Douglas type production model and the Weibull abatement function version of the damage

Table 1. Estimated Marginal Products Evaluated at the Geometric Mean Using Cobb-Douglas Type and Damage Control Models, U.S. Agriculture 1987

Input	Cobb-Douglas Type	Damage Control Model
Labor	44.54	46.53
Land and Buildings	0.04	0.04
Machinery	1.25	1.27
Other	1.29	1.29
Fertilizer	2.72	1.84
Pesticide	5.94	6.88

Source: Carrasco-Tauber and Moffitt (1992).

control model. Marginal products in the table are evaluated at the geometric means of the sample data variables. According to the Lichtenberg and Zilberman (1986) finding (Fig. 1), the marginal products using the different models should be the same at the geometric means. However, the Cobb-Douglas type model actually provides a lower estimated marginal product than that estimated with the damage control model (Table 1).

Even though sweeping econometric generalizations for the superiority of the damage control model relative to alternatives are not possible, use of the abatement function to encapsulate the pesticide variable(s) makes intuitive sense. Even greater intuitive appeal may be afforded specifications that extend the damage control model to account for the pest population and the notion of phytotoxicity. The next section focuses on extending the damage control model in both of these directions.

3. EXTENSIONS OF THE DAMAGE CONTROL SPECIFICATION

The original damage control specification is given as:

$$Q = F_1(Z) + F_2(Z)G(X) + \varepsilon \tag{1}$$

where

$$F_1(Z) = \text{minimum output} \tag{2}$$

$$F_1(Z) + F_2(Z) = \text{potential output} \tag{3}$$

and

$$0 \leq G(X) \leq 1 \tag{4}$$

Output is Q. The vector Z represents usual inputs, and the vector X represents inputs to control damage. When $G(X) = 1$, no damage occurs, and when $G(X) = 0$, the maximum damage occurs. G(X) gives the proportion of damage avoided by the control variable X. This model is illustrated in Fig. 2. Note that the initial pest infestation is not specified in the model.

In the case of pesticides, the damage control specification would account for varying infestation levels of the pest population, B, by including it as an argument in F_1 and G:

$$Q = F_1(Z, B) + F_2(Z)G(X, B) + \varepsilon \tag{5}$$

To explain Eq. (5), consider the simplest model with linear damage and a bug infestation equal to B_1, so $Q = Q_0 - \delta B_1$. Obviously, if enough pesticide is applied to kill the entire pest population, then output would equal Q_0, but

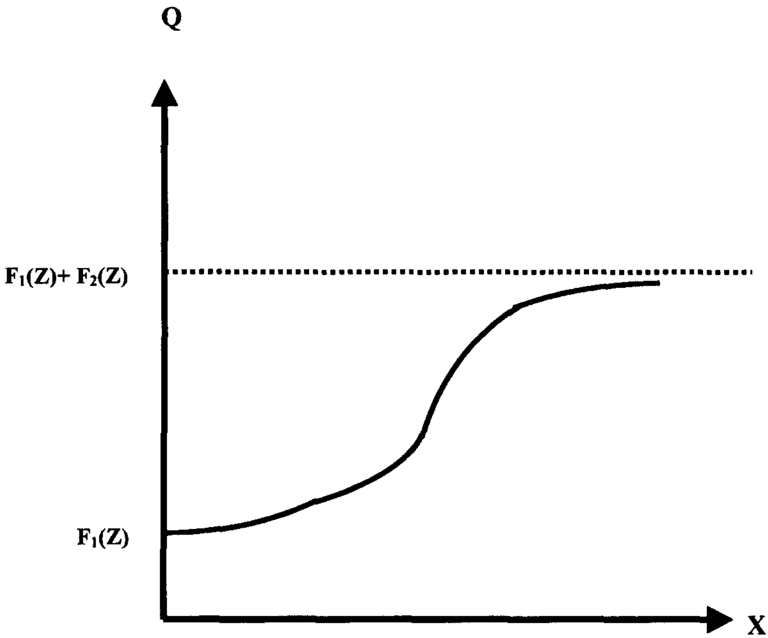

Fig. 2. Damage Control Specification.

without application of pesticide, output equals $Q_0 - \delta B_1$, shown in Fig. 3. Therefore, the output with zero pesticide application, F_1 in the damage control model, is a function of the initial infestation B. With a higher level of infestation, B_2, if no pesticide is applied then output is lower, equal to $Q_0 - \delta B_2$ as shown in Fig. 3. The proportion of damage avoided will in general depend not only on the amount of pesticide but also on the initial infestation. So, for example, if $X = X^*$ in Fig. 3, the proportion of damage depends on whether the initial bug population equals B_1 or B_2:

$$Q_1^* = F_1(Z, B_1) + F_2(Z)G(X^*, B_1) \tag{5a}$$

$$Q_2^* = F_1(Z, B_2) + F_2(Z)G(X^*, B_2) \tag{5b}$$

and the proportions of damage for the two initial infestations, B_1 and B_2, are $(Q_0 - Q_1^*)/Q_0$ and $(Q_0 - Q_2^*)/Q_0$.

When the initial bug infestation population is omitted in the econometric estimation, then it is included in the error term. For any IPM program (Hall & Duncan, 1984), where the quantity and timing of pesticide application awaits an infestation greater than the economic threshold (Headley, 1971; Hall &

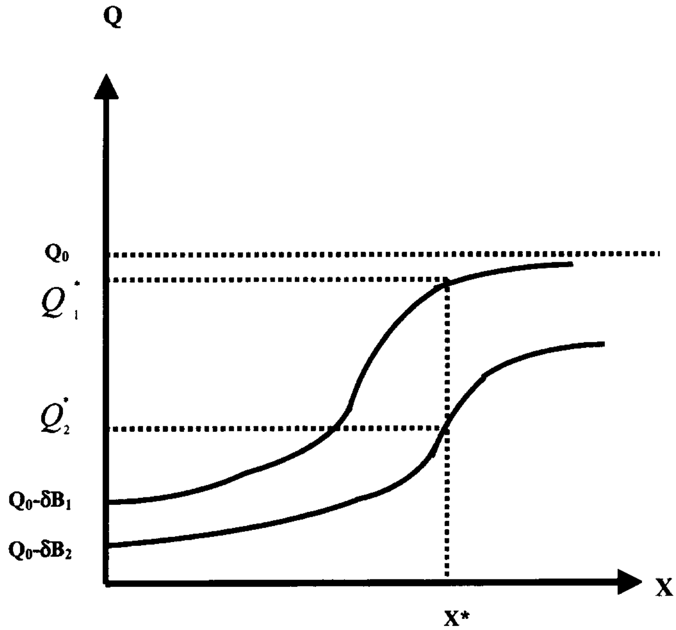

Fig. 3. Simplest Model with Linear Damage and $B_2 > B_1$.

Norgaard, 1973), the error term is correlated with the pesticide input X and the estimators are biased. Only when the pesticides are applied prophylactically, such as in a calendar-spraying program, would the bias be absent.

The solution to the problem of bias is to explicitly model the process generating the data, which must include the bug population. Rather than strictly continue with the damage control specification, we reformulate the problem. To do so, consider the motivation behind the approach. Output equals output without damage minus the damage. The damage equals output times the percentage change in damage, which depends on the infestation that survives the pesticide application, as shown in the Eq. (6) below. Equation (7) states that the percentage of the pest population killed depends on the amount of pesticide. Equation (8) constrains between zero and one both the percentage of damage and the percentage of the pest population killed by the pesticide.

$$Q = f(Z) - f(Z)H(B_A) \tag{6}$$

$$(B_B - B_A)/B_B = K(X) \tag{7}$$

$$0 \le K(X),\ H(B_A) \le 1 \tag{8}$$

To see how (5) relates to (6)–(8), solve (7) for B_A,

$$B_A = B_B[1 - K(X)] \tag{7a}$$

and substitute (7a) into (6) to obtain

$$Q = f(Z) - f(Z)H(B_B[1 - K(X)]) \tag{6a}$$

Since $0 \leq H(\cdot) \leq 1$, we see that f(Z) in (6a) equals $F_1(\cdot) + F_2(\cdot)$ in (5), the maximum possible output possible. From (6a), the maximum output possible occurs if either: (a) the initial infestation, B_B, is zero; or (b) the percentage killed is 100%, $K(X) = 1$, so that $H(0) = 0$. From (5), the maximum output occurs if $G(\cdot) = 1$, i.e. damage is zero. To model the process that generates the data, we have, therefore, substantially revised the damage control model.

To continue with this reformulation, let B_B and B_A be the initial infestation of the bug population (before application) and the surviving bug population (after application of pesticide). The most tractable function is the exponential. Specify the percentage of crop damaged and the percentage of the pest population killed as follows:

$$H(B_A) = [1 - \exp(-\delta B_A)] \tag{9}$$

$$K(X) = [1 - \exp(-\kappa X)] \tag{10}$$

Substituting these functional forms into the general expressions above, and adding multiplicative error terms results in the structural equations:

$$Q = f(Z)\exp(-\delta B_A + \varepsilon) \tag{11}$$

$$B_A = B_B \exp(-\kappa X + \eta) \tag{12}$$

In Eq. (11) above, as the surviving bug population after application goes to zero, output approaches f(Z), and as the surviving bug population approaches infinity, output goes to zero. Similarly for the kill function, as $X \to 0$, $B_A \to B_B$, and as $X \to \infty$, $B_A \to 0$.

Why reformulate the problem? First, the system of equations is linear after taking logs, so the estimation procedure is standard. Second, it is easy to further modify the model to account for phytotoxicity and retain a log-linear model, as shown below. Third, the original damage control specification was formulated for easy comparison to the Cobb-Douglas so the bias of historical studies could be easily recognized. That purpose was well served, but to make that comparison, Lichtenberg and Zilberman set $F_1(Z)$ to zero. From the original damage control formulation of the problem, $F_1(Z)$ has no interpretation except as the minimum output, whereas Fig. 3 shows that $F_1(Z)$ equals output (Q_0) if no infestation occurs minus the damage (δB) done by the infestation if left uncontrolled. Setting $F_1(Z)$ to zero implies that the initial infestation must be a very special population level – that level which would exactly destroy the entire crop and no more nor no less. By reformulating the problem we avoid the temptation to make such a peculiar assumption.

Fourth, in the original damage control specification, the interior solution guarantees that the marginal product of pesticides is always in Stage 2 of production, an undesirable assumption. It is desirable that the specification permit both Stage 1 and Stage 3, as well as Stage 2 of production. Different levels of initial pest infestation correspond to shifts in the marginal product curve for pesticides. Ceteris paribus, the model should allow for the possibility that at low levels of pesticide application, the marginal product curve rises with increasing amounts of pesticide. The shape of the marginal product curve at low doses relative to higher ones reflects the efficacy of the pesticide relative to the rate of damage caused by the remaining pest population. Phytotoxicity could cause the third stage of production to occur. Now let us check whether the extended damage control specification in Eqs (11) and (12) has these desirable properties.

Dropping the error terms for now, the reduced form is given by:

$$Q = f(Z)\exp[-\delta B_B \exp(-\kappa X)] \tag{13}$$

The marginal product of pesticides is given by:

$$MP_X = B_B f(Z)\delta\kappa \exp[-\kappa X - \delta B_B \exp(-\kappa X)] \tag{14}$$

Since the exponential function is everywhere positive, and so are B_B, δ, and κ, the marginal product curve is positive, which rules out Stage 3.

Before we further extend the model to allow for stage 3, we further consider the properties of the marginal product. The intercept of the marginal product occurs where $X=0$ and $MP_X = B_B f(Z)\ \delta\kappa \exp(-\delta B_B)$. The slope of the marginal product is given by:

$$\partial(MP_X)/\partial X = [-\kappa + \delta B_B \kappa \exp(-\kappa X)]MP_X \tag{15}$$

and this is greater than or less than zero depending on whether the term in [brackets] on the rhs is greater than or less than zero. As X approaches infinity, the term in brackets approaches $-\kappa$, so for sufficiently large X, the MP_X is negatively sloped, and therefore this function supports Stage 2 of production. At $X=0$, the expression in [brackets] is positive as long as $\delta B_B > 1$, which is determined by the data. So Stage 1 exists if supported by the data. The marginal product reaches a maximum where the slope equals zero, at $X^* = \{\ln[\delta B_B]\}/\kappa$, and at that point, $MP_X = \kappa f(Z)\exp(-1)$. Given the possible shapes of the marginal product of pesticides, the first order conditions for profit maximization are not sufficient for a maximum. Moreover, depending on pesticide prices, the corner solution $X^* = 0$ may maximize profit, quite apart from the corner solutions caused by fixed application costs or maximum legal doses (Hall, 1988).

To add in the phytotoxic effect of herbicides by allowing for stage 3 of the production function, simply let the percentage of damage to the crop depend on both the level of pest infestation, B_B, as well as the dosage of pesticide, X. The damage control specification is then further extended to:

$$Q = f(Z) - f(Z)H(B_A, X) \tag{16}$$

$$(B_B - B_A)/B_B = K(X) \tag{17}$$

$$0 \leq K(X), H(B_A, X) \leq 1 \tag{18}$$

For the special case of the user friendly exponential, we have estimable structural equations:

$$Q = f(Z) \exp(-\delta B_A - \varphi X + \varepsilon) \tag{19}$$

$$B_A = B_B \exp(-\kappa X + \eta) \tag{20}$$

The coefficient φ is the parameter that expresses phytotoxicity. For the moment, dropping the error terms gives the following reduced form:

$$Q = f(Z) \exp\{-\delta B_B[\exp(-\kappa X)] - \varphi X\} \tag{21}$$

Differentiating with respect to X gives the marginal product of pesticides,

$$MP_X = [\kappa\delta B_B \exp(-\kappa X) - \varphi]Q \tag{22}$$

This expression has the properties we want. The third stage of production occurs when the marginal product is negative, which occurs when

$$B_B\delta\kappa \exp(-\kappa X) < \varphi \tag{23}$$

Since the left-hand side of the inequality has the exponential function, which asymptotically approaches zero as the argument approaches minus infinity, for sufficiently large X the third stage of production occurs. In fact, setting the marginal product to zero, we can solve for the level of pesticide at which the third stage occurs:

$$X(\text{3rd Stage}) = [\ln(B_B\delta\kappa) - \ln(\varphi)]/\kappa \tag{24}$$

4. EMPIRICAL APPLICATION

The structural equations are written in log-linear form:

$$\ln Q = \ln[f(Z)] - \delta B_A - \varphi X + \varepsilon \tag{25}$$

$$\ln(B_A) = B_B - \kappa X + \eta \tag{26}$$

If f(Z) is the Cobb-Douglas, or a more general power function (de Janvry, 1972), then this is a system of equations which is linear in the parameters to be estimated. If data are available from farms, then standard procedures for system estimation are appropriate.

For the application at hand, the data are from a controlled experiment (Hall, 1988). The variables for which measures exist are the log of yield (ln Q), pesticide dose (X), the insect population after application (B_A) and the log of the insect population (ln B_A); there are no measures of the variables Z, nor is there a measure of B_B. With substitution of coefficients for the parameters, the model becomes:

$$\ln Q = \beta_0 + \beta_1 B_A + \beta_2 X + \varepsilon \tag{27}$$

$$\ln B_A = \beta_3 + \beta_4 X + \eta \tag{28}$$

where β_2 is zero if there is no phytotoxic effect of the pesticide, and β_3 is an estimator for the initial infestation.

A random plot design generated the data, with varying amounts of pesticide application determined by experimental design (Hall, 1988). Consequently, for this application, two stage least squares is appropriate. Single equation estimation is acceptable for Eq. (28). Ordinary least squares gives estimates for β_3 and β_4 which in turn generate predicted values for ln B_A and B_A. Equation (27) is then estimated using the predicted values of B_A as an instrument for B_A.

Tables 2 and 3 show the results of the estimation. For Eq. (27), called the damage equation in Table 2, none of the coefficients are significant. Since the pesticide is an insecticide, not a herbicide, it is reasonable to drop the phytotoxicity term from the first equation by setting β_2 equal to zero, and re-

Table 2. Damage Equation (27) with Stage 3 (Phytotoxicity).

Variable	Coefficient	Std. error	T-Stat	2-Tail Sig.
β_0	−0.7810256	5.8260843	−0.1340567	0.8939
β_1	0.0097632	0.0377757	0.2584514	0.7971
β_2	0.6444547	2.1863014	0.2947694	0.7694
Mean of ln Q	0.778575		F-statistic	0.220722
S.D. of ln Q	0.109926	N = 52	Prob(F-statistic)	0.802733
S.E. of regression	0.522563		Σe^2	13.38053

Table 3. Pesticide Efficacy Equation (28).

Variable	Coefficient	Std. error	T-Stat	2-Tail Sig.
β_3	5.1002750	0.1008020	50.596946	0.0000
β_4	−0.7573642	0.1158406	−6.5379877	0.0000
Mean of ln Q	4.590511		F-statistic	42.74528
S.D. of ln Q	0.621281	N = 52	Prob(F-statistic)	0.000000
S.E. of regression	0.460709		Σe^2	10.61265

Table 4. Damage Equation (27) Without Stage 3 (Without Phytotoxicity: $\beta_2 = 0$).

Variable	Coefficient	Std. error	T-Stat	2-Tail Sig.
β_0	0.9565107	0.0603293	15.854826	0.0000
β_1	−0.0015431	0.0005056	−3.0520837	0.0036
Mean of ln Q	0.778575		F-statistic	9.315215
S.D. of ln Q	0.109926	N = 52	Prob(F-statistic)	0.003634
S.E. of regression	0.111893		Σe^2	0.625997

estimate Eq. (27). When the phytotoxicity term is dropped and the damage equation is re-estimated, the results shown in Table 4 are significant for the damage equation; in this case the empirical results rule out Stage 3 of production.

IV. CONCLUDING REMARKS

A rigorous econometric rationale for choosing the damage control model over other empirical models for pesticide studies is not evident. Even so, it may be intuitively appealing to use the damage control model and, if it is used, an extension which permits additional flexibility seems warranted.

The extended damage control specification for pesticide productivity, as developed above, allows the data to determine whether Stages 1 and 3 of production exist, rather than assuming those stages of production do not exist. The corner solution of zero application is possible in the extended damage control specification so that organic farming (Hall et al., 1989) can be explained within the context of the model.

It is noteworthy that special studies may be needed for pesticide productivity estimates since, for example, United States Department of Agriculture and perhaps other official surveys of growers typically don't collect pest infestation information (except for subjective ratings by the respondents in some cases). Hence, controlled experiments or other special studies where scientists or independent pest control advisers measure pest infestations may be necessary.

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3. INDUCED INNOVATION AND THE ECONOMICS OF HERBICIDE USE

Jorge Fernandez-Cornejo and Yvan Pho

ABSTRACT

We present direct econometric tests of the induced innovation hypothesis. We test whether the price of herbicides relative to labor, machinery, and land, as well as research stocks, affects the direction of technological change and long-run substitution of herbicides for labor, machinery, and land, in U.S. agriculture. In the long run, a decrease in the price of herbicides relative to labor induces a strong labor-saving and herbicide-using bias in technological change. Public research induces labor-saving, machinery-saving, land-saving, and herbicide-using biases. Exogenous changes in scientific knowledge and/or spillovers from other sectors are labor and machinery saving and herbicide using.

Wide use of herbicides or chemical "weed killers" began in the mid-1940s, after the discovery of effective synthetic herbicides such as 2,4-D. U.S. herbicide production grew from 2,000 tons in 1945 to 280,000 tons in 1975. By the early 1980s, herbicides had substantially replaced mechanical or manual methods of weed control contributing to greater efficiency and productivity in

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U.S. agriculture. For example, about 6 labor-hours were required to grow and harvest one acre of corn yielding 33 bushels in 1945, when only 5% of the corn acres were treated with herbicides. In comparison, half as many labor hours were required per acre to yield almost three times as many bushels of corn (88 bushels) in 1975, when about 90% of the corn acres were treated with herbicides.

As Green et al. (1977) observe, it is easier to examine the economics of herbicide usage compared to the economics of insecticides and fungicides, "as herbicides are essentially used as a substitute to mechanical or hand weeding and can be assessed on this basis." Just as capital equipment (machinery) substituted for labor in U.S. agriculture (as the price of machinery relative to labor decreased) herbicides may have substituted for both labor and machinery, contributing to the increase in the herbicide/labor and herbicide/machinery ratios. For example, a farmer that experiences a decrease in the price of herbicides relative to labor has a choice of substituting herbicides for labor, applying herbicides to the weeds rather than hand-picking the weeds or hand hoeing. Similarly, if the price of herbicides decreases relative to the price of machinery, herbicides may be used to replace machinery (for example, to displace mechanical tillage).[1]

The induced innovation hypothesis, as presented by Hayami and Ruttan (1985), examines the long term linkages between relative input prices, research and development, and the development of new technologies which allows the substitution of abundant for scarce inputs. In simple terms, the theory argues that "technological change responds to price movements so as to save on factors of production that have become relatively more expensive" (Fulginiti, 1994).

While early empirical applications of the theory of induced innovation (Hayami & Ruttan, 1977; Binswanger, 1978) have been useful to study historical trends in agriculture, very few econometric studies specify prices explicitly as determinants of technical change and factor bias (Frisvold, 1991; Fulginiti, 1994). As observed by Frisvold (1991), previously used indirect tests did not specify directly factor prices as determinants of factor biases but rather they measured factor biases as functions of a time trend and compared this trended bias with relative price movements.[2] The indirect method has several flaws, including omitted variable econometric bias and only yields statistically meaningful results when the null hypothesis of induced innovation is false.

Another problem with most previous empirical testing of the induced innovation hypothesis is the use of input prices and quantities unadjusted for quality. As Kislev and Peterson (1981) observe, quality-unadjusted inputs are not only inconsistent but also meaningless in the analysis of technical change.

This study presents a direct test of the of induced innovation hypothesis by explicit econometric estimation of the long-run impact of the relative prices (of herbicides to labor, herbicides to machinery, herbicides to land), as well as research and development stocks, on the direction of technical change and long-run substitution among herbicides, labor, machinery, and land in U.S. agriculture. In addition, this paper uses quality-adjusted input prices and quantities in the empirical estimation and direct testing of the induced innovation hypothesis.

THE THEORETICAL FRAMEWORK

The induced innovation hypothesis is associated with John Hicks (1932), who asserted that new technologies are developed and adopted to save those factors that have become relatively more expensive. To illustrate, using the familiar input space, movements along an isoquant reflect the short-term responses to changes in relative prices. On the other hand, long term responses to changes in relative prices leads to changes in relative input use due to shifts of the isoquant induced by technical change. The envelope of all possible new isoquants resulting from technical change that can be attained by a given research budget is called the innovation possibility curve (Ahmad, 1966). As different technological choices are made, long run input substitution can take place along the innovation possibilities curve in a similar way to short run substitution, which takes place along the isoquant.

Following the development presented by Frisvold (1991), X_t, the vector of factor ratios at time t is a function of the current price vector P_t and a vector T that represents the current state of technology. T is a function of overall past research investments (B), past budget allocations (which depend on past input price expectations), and past scientific knowledge. To illustrate, consider only two periods t and $t-1$. Then, the current factor ratios can be expressed as: $X_t = X_t\,(P_t, P_{t-1}, B_{t-1}, t)$. Taking the total derivative of the ith component of X with respect to time yields:

$$\frac{dX_{it}}{dt} = \left[\frac{\partial X_{it}}{\partial P_t}\frac{dP_t}{dt}\right] + \left[\frac{\partial X_{it}}{\partial t} + \frac{\partial X_{it}}{\partial B_{t-1}}\frac{dB_{t-1}}{dt} + \frac{\partial X_{it}}{\partial P_{t-1}}\frac{dP_{t-1}}{dt}\right]$$

As noted by Frisvold (1991), the first term in brackets represents simple factor substitution while the second term in brackets is the factor bias, accounting for all changes in X_t, excluding changes in P_t.

THE EMPIRICAL MODEL

The model used is an extension of the framework developed by deJanvry et al. (1989) and empirically implemented by Frisvold (1991). The model uses a two-level, nested, constant elasticity of substitution production function (Karagiannis & Furtan, 1990) and includes five inputs, labor, machinery, land, fertilizers, and herbicides:

$$Q = \{\gamma[\beta_1(L)^{-\rho_1} + \beta_2(E_m M)^{-\rho_1} + (1-\beta_1-\beta_2)(E_h H)^{-\rho_1}]^{(-1/\rho_1)}$$
$$+ (1-\gamma)[\alpha_1(A)^{-\rho_2} + \alpha_2(E_f F)^{-\rho_2} + (1-\alpha_1-\alpha_2)(E_h H)^{-\rho_2}]^{(-1/\rho_2)}\}^{(-1/\rho)}$$

where Q, H, M, L, A, and F are quality-adjusted quantity indices of output, herbicide, machinery, labor, land, and fertilizer respectively; ρ, ρ_1, ρ_2 are the substitution parameters, and α, β and γ are distribution parameters. The input parameters E_m, E_f, and E_h are functions of past public and private research and development investments as well as the distribution of those investments among research areas (Frisvold, 1991). In turn, these parameters are a function of expected input prices.

Farmers are assumed to maximize profits (B) subject to an expenditure constraint K such as credit rationing (Lee & Chambers, 1986):

$$B = Max\{PQ - C(P_l, P_m, P_h, P_a, P_f; P, Q): C(P_l, P_m, P_h, P_a, P_f; P, Q) \leq K)\}$$

where P, P_h, P_m, P_h, P_a, and P_f are quality adjusted price indices for output, herbicide, machinery, labor, land, and fertilizer, respectively. The cost function is:

$$C(P_l, P_m, P_h, P_a, P_f; P, Q)$$
$$= Min\{(L\,P_l + MP_m + HP_h + AP_a + FP_f): (L, M, H, A, F, Q) \in T\}$$

The optimal factor ratios, are determined solving the above constrained maximization problem as a function of current price ratios, and the current state of technology (vector T) which is a function of overall past research investments, past budget allocations, and past scientific knowledge. Past budget allocations among research areas depend on expected input prices. The reduced-form equations are:

$$ln(H/L)_t = a_0 + a_1\, ln(P_h/P_l)_t + a_2\, ln[lag(P_h/P_l)_t] + a_3\, ln[lag(P_h/P_m)_t]$$
$$+ a_4\, ln[lag(P_h/P_a)_t] + a_5\, ln[lag(P_m/P_l)_t] + a_6\, ln[lag(P_f/P_a)_t]$$
$$+ a_7\, ln[lag(P_a/P_l)_t] + a_8\, ln(B)_t + a_9\, ln(R)_t + a_{10}t \qquad (1)$$

$$ln(H/M)_t = b_0 + b_1 ln(P_h/P_m)_t + b_2\ ln[lag(P_h/P_m)_t] + b_3\ ln[lag(P_h/P_l)_t]$$
$$+ b_4\ ln[lag(P_h/P_a)_t + b_5\ ln[lag(P_m/P_l)_t] + b_6\ ln[lag(P_f/P_a)_t]$$
$$+ b_7\ ln[lag(P_d/P_l)_t] + b_8\ ln(B)_t + b_9\ ln(R)_t + b_{10}t \quad (2)$$
$$ln(H/A)_t = c_0 + c_1\ ln(P_h/P_a)_t + c_2\ ln[lag(P_h/P_a)_t] + c_3\ ln[lag(P_h/P_m)_t]$$
$$+ c_4\ ln[lag(P_h/P_l)_t] + c_5\ ln[lag(P_m/P_l)_t] + c_6\ ln[lag(P_f/P_a)_t]$$
$$+ c_7\ ln[lag(P_d/P_l)_t] + c_c\ ln(B)_t + c_9\ ln(R)_t + a_{10}t \quad (3)$$

where B and R are the stocks of public and private research, and t stands for a time trend that represents "fundamental" biases in technological change (Bingswanger, 1974) and may result from "exogenous changes in scientific knowledge or technological spillovers from other industries" (Frisvold, 1991). As proxies for expected prices we use moving averages of past price ratios denoted above as lag(P_i/P_j). Finally, the term *ln* denotes natural logs.

DATA AND ESTIMATION

The model is estimated using time series data of U.S. agriculture. Quality-adjusted input price and quantity indices for the years 1948–1994 for labor services (hired and self-employed), machinery (capital services from durable equipment), land, and fertilizers are obtained from Ball et al. (1997). When necessary, output and inputs are aggregated using Tornqvist-Divisia indices.

Quality-adjusted indices for herbicides are updated for this study following Fernandez and Jans (1995). Adjusting herbicide quantity and price indices for quality is particularly important because of rapid technical change, as new and better herbicides are introduced and other products are banned or are dropped by their manufacturers because of health, environmental, or economic considerations.

The term herbicide refers to a very large number of heterogeneous products. Thousands of formulations (commercial forms in which the herbicides are sold) are used. These formulations are mixtures of active chemicals (active ingredients) and inert materials, which are used to improve safety and facilitate storage, handling, or application. Dozens of chemical products are used as herbicide active ingredients. Each active ingredient has a different potency, i.e. it must be applied at a different rate for a given degree of weed control, and has a different impact on human health and the environment. Given this heterogeneity, it does not seem appropriate to compute total herbicide use by adding the quantities of all herbicides used, even if expressed in the same units (e.g. pounds) of active ingredient (a.i.). For example, less than a tenth of a pound of a newer, more potent synthetic sulfonylurea herbicide has about the same degree of weed control as several pounds of older, weaker, herbicides such as EPTC or metolachlor. Calculation of quality adjusted price indices is based on hedonic methods in which herbicides are viewed as a bundle of qualities or characteristics which contribute to the productivity or utility derived from its use. Appendix I provides a summary of the method used to estimate the quality-adjusted herbicide series.

Annual research investments are used to calculate stocks of public and private agricultural research, as economists have recognized that research expenditures do not affect agricultural production immediately. Long lags between research expenditures and development and adoption of new technologies based on research discoveries have been reported and, in addition, the results of research will usually affect agricultural production for a long time. Thus, the effect of annual research expenditures on the development of new technologies is minimal in the beginning years, peaks in the mid-years, and declines in later years. For this reason, most recent economic studies of technological innovations consider the stock of research, rather than annual research expenditures as proxies for research output.

Research stocks are calculated using the principles commonly used in calculations of capital stocks (Huffman & Evenson, 1993; Chavas & Cox, 1992). Thus, a research stock variable (B_t) at year t is calculated as a function of past annual investments in research (b_{t-j}) made in years $t-j$ (where $j=1, 2, \ldots, m$ and m is the maximum number of lags) from the following expression (Chavas & Cox, 1992):

$$B_t = \sum_{j=l}^{m} a_j b_{t-j}$$

Where the parameter $a_j \geq 0$ measures the marginal impact of research conducted at time $t-j$ on the research stock.

Following Chavas and Cox (1993), we consider a maximum lag of 30 years for both private and public research and development. The marginal impact of research (parameter a_j) was also obtained from Chavas and Cox (1993). It is important to note that while public and private research have similar maximum lags, their marginal impact is quite different. The marginal impact of private research peaks at about the 15th year and drops rapidly to become negligible at the 22nd year while the marginal impact of public research is minimal for the first 15 years and reaches a maximum at the 22nd year.

Given the long lags between research investment and technical change, we calculate research stocks using annual data on research investments in U.S.

agriculture for more than 100 years, from 1889–1994. Research stocks are computed for the following categories: public sector expenditures (made by USDA, State Agricultural Experimental Stations, and total) and private sector expenditures. Annual public investments on agricultural research are obtained from Huffman and Evenson (1993, pp. 95–96), Alston and Pardey (1996, p. 76), and the Current Research Information System (USDA, various years). Annual private research expenditures are from Huffman and Evenson (1993, pp. 95–96) and Klotz et al. (1995).

The Eqs (1)–(3) are estimated together using the data in an iterated seemingly unrelated regression (ITSUR) framework (Zellner, 1962). After model estimation and statistical testing, the short and long-run impact of relative prices and the long-run effect of private and public research stocks on the herbicide/labor, herbicide/machinery, and herbicide/land ratios are measured in elasticity form.

RESULTS

Figure 1 shows the quality-adjusted quantity indices for herbicides, machinery, labor, and land used in U.S. agriculture between 1948 and 1994. As herbicide use began during World War II and only 5% of the corn acres used herbicides

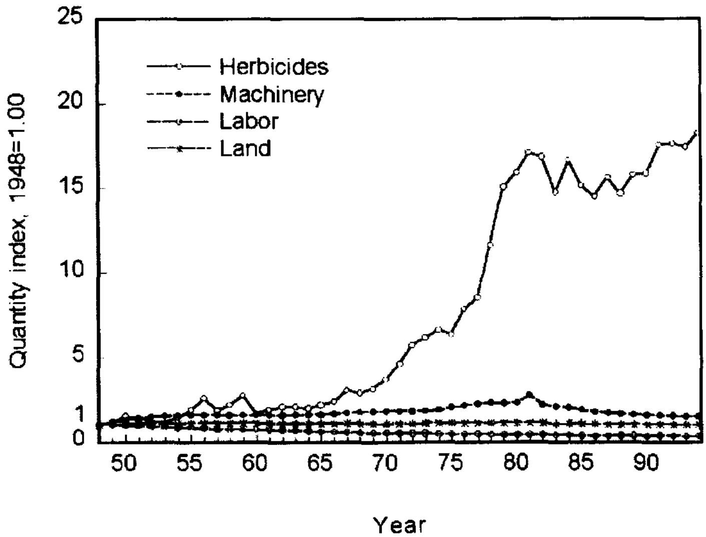

Fig. 1. Quantity-Adjusted Input Indices for U.S. Agriculture.

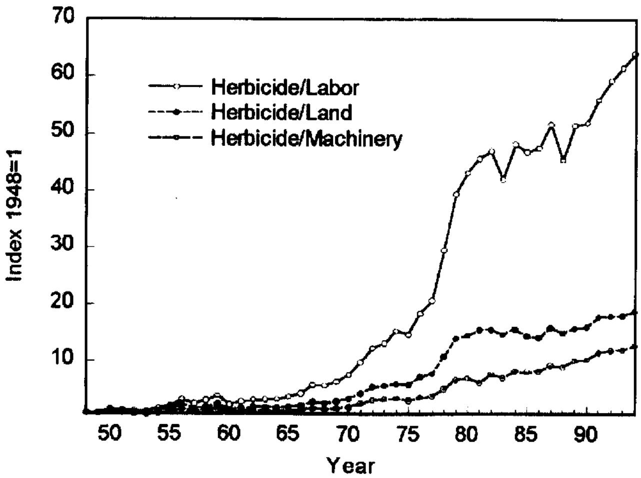

Fig. 2. Ratios of Quality-adjusted Indices of Input Quantities.

in 1945, Fig. 1 really captures most of the evolution of the growth cycle for herbicides. As seen in Fig. 1, quality-adjusted herbicide use rose 18 fold between 1948 and 1981 and stabilized during the 1980s and early 1990s, as the mature portion of the cycle was reached.

Quality-adjusted labor use decreased continuously during the period, reaching in 1994 about a third of the 1948 labor use, while quality-adjusted machinery use peaked in 1981 at 2.7 times the 1948 usage and decreased thereafter, reaching in 1994 a level only 45% higher than in 1948. As a consequence, herbicide/labor ratios rose more than 60 fold between 1948 and 1994 while herbicide/machinery ratio increased about 15 times, and the herbicide/land ratio rose about 20 fold in the same period (Fig. 2).

Figure 3 shows that the dramatic rise of the quality-adjusted herbicide/labor ratio was accompanied by a drop in the quality-adjusted herbicide/labor price ratio, which reached in 1994 about one third of the 1948 price. Similarly, Fig. 4 presents the change of the herbicide/machinery price ratio. Note, however, that the decline of the herbicide/machinery price ratio is not uniform. After declining during most of the fifties, it increased during the sixties, peaking in 1968 and then again in 1975. The herbicide-machinery price ratio dropped between 1975 and 1983 and rose slightly the rest of the period.

Table 1 shows the ITSUR regression results. Overall fit is good, measured by the adjusted R-squared for all equations. The coefficients of the current price ratios represent the short run direct elasticities of substitution. The short-run

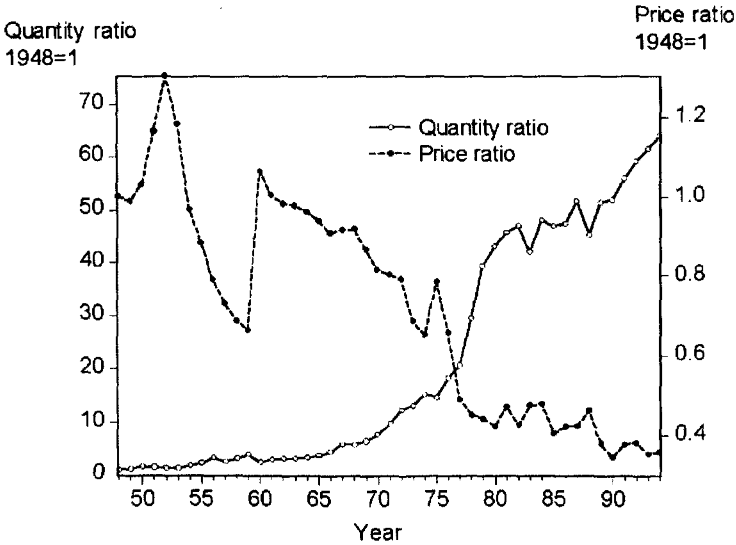

Fig. 3. Herbicide/Labor Quantity and Price Ratios in U.S. Agriculture.

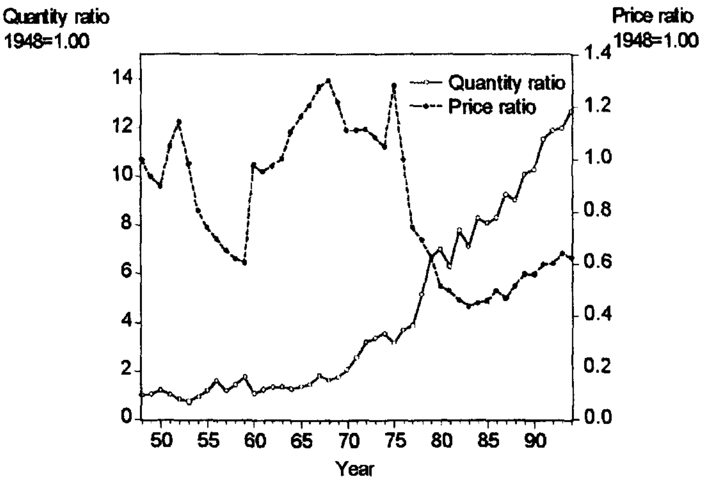

Fig. 4. Herbicide/Machinery Quantity and Price Ratios in U.S. Agriculture.

Table 1. ITSUR Parameter Estimates for Factor Ratio Equations – U.S. Agriculture, 1948–1995.

	Herbicide/labor ratio[1]		Herbicides/machinery ratio[1]		Herbicides/land ratio[1]	
	Parameter	t-ratio	Parameter	t-ratio	Parameter	t-ratio
Intercept	−3.76***	−12.6	−2.96***	−9.48	−2.76***	−9.56
Herbicide/labor price ratio	−0.23***	−3.89				
Herbicide/machinery price ratio			0.17*	2.00		
Herbicide/land price ratio	−0.004	−0.17				
Lagged (Herbicide/labor price ratio)	−13.51**	−2.32	−11.80*	−1.90	−16.13***	2.76
Lagged (Herbicide/machinery price ratio)	12.85**	2.11	10.50	1.62	15.39**	2.51
Lagged (Herbicide/land price ratio)	0.74	0.64	1.53	1.26	0.93	0.80
Lagged (Machinery/labor price ratio)	11.61*	1.92	10.02	1.56	14.29**	2.35
Lagged (Fertilizer/land price ratio)	0.64***	2.77	0.18	0.72	0.53**	2.31
Lagged (Land/labor price ratio)	2.03*	1.89	2.03*	1.78	1.99*	1.85
Stock of public research	1.16***	3.34	1.48***	3.99	1.41***	4.05
Stock of private research	−1.45**	−2.62	−2.19***	−3.67	−1.92***	−3.49
Time trend	0.11**	2.17	0.16***	2.78	0.13**	2.44
Adjusted R^2	0.99		0.98		0.98	

[1] All variables (left and right hand side), except the time trend, are expressed in logs.
*, **, *** = Statistically significant at the 10, 5, and 1% level, respectively.

elasticity for the herbicides/labor ratio is very significant and negative as expected (–0.23). This means that a decrease of 1% in the quality-adjusted price ratio of herbicides to labor (holding all other prices constant) led in the short run to an increase of 0.23% in the quantity ratio of herbicides to labor.

The short run elasticity for the herbicide/land ratio is also negative but not statistically significant, while the short-run elasticity for the herbicide/ machinery substitution is positive (0.17). The herbicide/machinery result may be due to the machinery data series. Unlike the fertilizer, and labor series that were fully adjusted for quality by Ball et al., it appears that the machinery series were only partially adjusted for quality. This would also explain the behavior of the herbicide-machinery price ratio (Fig. 4). If a price time series is not adjusted for quality, or it is only partially adjusted, the price will increase with time at a faster rate than a fully quality-adjusted price series. Thus, a herbicide/machinery price series in which the machinery prices were not fully adjusted for quality will decline at a slower rate compared to a series fully adjusted for quality. This situation may cause an apparent violation of economic theory, as the herbicide/machinery ratio increases when the herbicide/machinery price ratio apparently is also raising. In fact, however, the increase in the herbicide/machinery ratio may be an economically consistent response to an actual *decline* in the quality-adjusted herbicide/machinery price ratio. Another reason for these findings may be that substitution of herbicides for machinery has also resulted from forces beyond herbicide/machinery prices; for example, the rapid adoption of conservation tillage to control soil erosion.

Results for the coefficients of the expected price ratios indicate that in the long run a decrease in the herbicide/labor price ratio induces a strong labor-saving and herbicide-using bias, as expected from the induced innovation hypothesis. The long-run elasticity of the herbicide/labor ratio with respect to the ratio of the respective prices, calculated at the means, is –13.51. This means that a decrease of 1% in the expected price ratio of herbicides to labor led in the long run to an increase in the quantity ratio of herbicides to labor of 13.5%. While this elasticity appears to be quite high, note that it only applies in the long run. From this perspective, the elasticity is consistent with Fig. 3 that shows that the herbicide/labor quantity ratio increased 60 fold while the respective price ratios dropped about a third.

As expected, increases in investment in public research induced a labor-saving and herbicide using bias in technological change. The elasticity of herbicide/labor ratio relative to stock of public research is + 1.16, meaning that a 1% increase in the stock of public research increases the herbicide/machinery ratio by 1.16%. Similarly, increases in investment in public research induced a machinery-saving and herbicide-using bias. The elasticity of herbicide/ machinery ratio relative to the stock of public research is +1.48. Finally, increases in investment in public research induced a land-saving and herbicide-using bias. The elasticity of herbicide/land ratio relative to the stock of public research is +1.41.

However, contrary to our original expectation, investment in private research induced a labor and machinery-using and herbicide-saving bias. One possible explanation for this finding is related to weaknesses in the private research data, which is not as complete as the public research series due to the many sources from which it is necessarily being collected. In addition, the data on private research expenditures include several components that are not relevant to agriculture viewed from our rather narrow perspective (crops). More importantly, the proportion of these "extraneous" components varies from year to year causing some distorting trends in the data. For example, the percent of the private research investments devoted to "food and kindred products" declined from 45% to 30% in 1992, while private research investments in "animal health" increased from 6% in 1960 to 9% in 1992.

The coefficients of the time trend indicate that there is a fundamental bias towards labor- saving and herbicide-using (elasticity = 0.11), machinery-saving and herbicide-using (elasticity = 0.16), and land-saving herbicide-using (elasticity = 0.13) technological change. This is an indication that exogenous changes in scientific knowledge and/or spillovers from other sectors are labor and machinery saving and herbicide using. In addition, the trend may be picking up those effects of the private research not fully accounted in the corresponding terms because of the noted weaknesses of the private research data.

CONCLUDING COMMENTS

This paper presents a direct test of the induced innovation hypothesis by explicit econometric estimation of the long-run impact of the relative prices (of herbicides to labor, herbicides to machinery, and herbicides to land), as well as research and development stocks, on the direction of technical change and long-run substitution among herbicides, labor, machinery, and land in U.S. agriculture. In addition, this paper uses quality-adjusted input prices and quantities in the empirical estimation and testing.

Most of the results agree with our expectation. Herbicide is a short run substitute for labor. In the long run, a decrease in the price of herbicides relative to labor induces a strong labor-saving and herbicide-using bias in technological change. However, a decrease in the herbicide/machinery price ratio did not lead to a significant machinery-saving bias, apparently because of the incompletely adjusted machinery data series.

As expected, increases in investment in public research have induced labor-saving and herbicide-using biases in technological change. Increases in investment in public research have also induced machinery-saving and herbicide-using biases, and land-saving and herbicide-using. However, contrary to our expectation, private research investment appears to have caused herbicide-saving and labor, and machinery-using biases. A possible reason for this result may be weaknesses in the private research data.

The coefficients of the time trend indicate that there is a fundamental bias towards labor-saving/herbicide using; machinery-saving/herbicide using; and land-saving/herbicide using technological change. This suggests that exogenous changes in scientific knowledge and/or spillovers from other sectors are labor and machinery saving and herbicide using.

The findings of this study highlight the importance of market forces not only to effect factor substitution within the constraints of the technology available but also in the very long run, via technical change. Thus, input price policies (particularly used in developing countries) have important implications beyond what is commonly acknowledged.

NOTES

1. This is simplified situation, given that machinery may also be used to improve herbicide applications. In these cases, labor may be a complementary input with both machinery and herbicides. Moreover, substitution of herbicides for machinery has also taken place as a result of forces other than herbicide/machinery prices; for example, the adoption of conservation tillage to control soil erosion.
2. Technical change is said to be biased, as opposed to neutral, when it results on changes in factor proportions.
3. The toxicity data required by the EPA for each chemical pesticide (active ingredient) include the results of a large number of different types of tests, including acute oral, dermal, and inhalation studies, a two-generation reproduction study, chronic feeding studies on rodents and nonrodents, teratogenicity studies on rats and rabbits, oncogenicity studies in mice and rats, mutagenicity studies, and delayed neuropathy studies.

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APPENDIX

Estimation of Quality-adjusted Herbicide

This appendix discusses empirical issues related to the estimation of quality-adjusted price and quantity indices for herbicides; more detail is given in Fernandez-Cornejo and Jans (1995). The calculation of quality-adjusted price indices is based on hedonic methods in which a commodity is viewed as a bundle of qualities or characteristics that contribute to the productivity or utility derived from its use. According to the hedonic framework (Rosen, 1974; Triplett, 1989) the price of a commodity represents the valuation of the "characteristics that are bundled in it," and each characteristic is valued by its "implicit" price. Implicit prices for characteristics exhibit many of the properties of ordinary prices but are seldom observed directly and must be estimated from the hedonic function (Triplet, 1989). Griliches (1964) noted that if we can observe different "quality combinations" selling at different prices, it is possible to estimate, at the margin, the price of these qualities.

A herbicide hedonic function may be expressed as $P = f(X, D)$; where P represents herbicide prices, in dollars per pound a.i.; X is the vector of characteristics or "quality" variables, and D is a vector of other variables. We use as quality characteristics herbicide potency, a toxicity index, and a measure of the persistence of the herbicide in the environment. Because herbicide application rates (in pounds a.i. per acre) are inversely related to herbicide potency, rates are used as a measure of the potency of each a.i. to protect a given crop. It is expected that a more potent herbicide (used at a lower rate to achieve a given degree of pest control than a weaker herbicide) should command a higher (per pound) price, *ceteris paribus*. In consequence, herbicide prices should be inversely related to rates.

The toxicity index that we use was developed by Fernandez-Cornejo and Jans (1995) and encompasses acute and chronic toxicity for humans (or mammals in general) and summarizes the results of a large number of measures of mammalian toxicity.[3] The toxicity index is a modification of the "combined risk index" proposed by the RAND Corporation (Hammitt, 1986) and can be expressed in a logarithmic scale as LOTI = (LAI + LCI + LTI)/3, where LAI is the acute element in the index, LCI is the cancer-related chronic element, and LTI is the teratogenic component. Regarding persistance, we included it in the hedonic function as a dummy variable that divides all herbicides into two groups, high (half lives of more than 60 days) and low persistence.

After allowing for all major differences in product characteristics by holding them constant through regression techniques, the part of the price change not accounted for by the included characteristics will be reflected in the time dummy coefficients; this is our best estimate of the price change unexplained by changes in product characteristics (Griliches, 1964). Introduction of variables other than quality, such as the price of the crop or patent dummies, is useful if the main objective of the study is to determine implicit prices; but it may bias the adjusted price indices.

While some empirical hedonic studies have preferred the use of the semilog or log-log forms, the functional form of the hedonic function is entirely an empirical matter (Triplett, 1989). For this reason we use the Box-Cox statistical procedure (Cropper et al., 1988) to select the most appropriate functional form of the hedonic function. A linear Box-Cox function is specified following Cropper et al. (1988), who recommend this form for hedonic price functions which often make use of proxies for "hard to measure attributes" and show that the linear Box-Cox form "consistently outperforms the quadratic Box-Cox." Our regression model is:

$$P(\lambda_1) = X(\lambda_2)\beta + D\gamma + \varepsilon \tag{1}$$

where $P(\lambda_1)$ is the Box-Cox transformation of the dependent variable price, $P > 0$; i.e.

$$P(\lambda_1) = \begin{cases} \dfrac{P^{\lambda_1} - 1}{\ln P^{\lambda_1}} & \begin{matrix} if \quad \lambda_1 \neq 0 \\ if \quad \lambda_1 = 0 \end{matrix} \end{cases} \tag{2}$$

Similarly, $X_j(\lambda_2)$ is the Box-Cox transformation of the continuous quality variables $X_j > 0$ (for $j = 1,2$); i.e. $X_j(\lambda_2) = (X_j^{\lambda_2} - 1)/\lambda_2$ if $\lambda_2 \neq 0$ and $X_j(\lambda_2) = ln\ X_j$ if $\lambda_2 = 0$. D is a vector of dummy variables, not subject to transformation; β and γ represent unknown parameter vectors, and ε, is the disturbance term. Several hedonic functions are evaluated, corresponding to the herbicides used in the four major crops, (corn, cotton, sorghum, and soybeans).

Several methods have been used to calculate price changes adjusted for quality using hedonic functions, including the imputation, characteristics price, and dummy variable techniques. The latter is used because it is simpler, and Triplett (1989) has provided extensive empirical evidence of the robustness of the hedonic price indices to the calculation method. Using the dummy variable technique, quality adjusted price changes are calculated directly from the coefficients of the time dummy variables (D) in hedonic regressions such as Eq. (1) run on pooled cross-sections of two or more years. The price indices can be obtained from one regression run over the sample period, from a series of "adjacent years" regressions, or from a series of overlapping subperiods of three or more years. To the extent that (imputed) prices of quality characteristics change through time, the series of two-year regressions would be preferable because it allows the slopes β to change through time (Gordon, 1990), but the advantages of the adjacent-year regressions are partially offset by the reduced sample size for each regression and the possibility that the coefficients may move erratically from year to year (Nelson et al., 1994)

While there is no theoretical reason why the coefficients should vary or should remain constant through time, we presume that the implicit prices of some of the characteristics have changed within the period of analysis. Concern over health and environmental issues by farmers and the public has become increasingly important in recent years. As a result, the implicit price of "a unit of toxicity" may have risen (in absolute value) with time. To examine if the data supports our expectation of changing parameters (and if so when have those changes occurred) we conducted a series of Chow tests and pooled one or more groups of adjacent years only if the results indicate that the coefficients are stable within such groups of adjacent years. The null hypothesis that the hedonic coefficients are stable with time is that the slope parameters are equal during the whole period analyzed. The alternative hypothesis is that the slope parameters vary from period to period.

The final hedonic functions with the time dummy variables appended are:

$$Ln\ P = \beta_0 + \beta_1 \frac{(X_1^{\lambda_2} - 1)}{\lambda_2} + \beta_2 \frac{(X_2^{\lambda_2} - 1)}{\lambda_2} + \gamma_p D_p + \sum_t \gamma_t D_t + \varepsilon \qquad (3)$$

where X_1 represent the potency, X_2 the toxicity index, D_p the herbicide persistence, and D_t the time dummies starting with the base year ($D_t = 1$ for year $= t$ and 0 otherwise). The coefficients of the time dummy variables may be interpreted as the estimated change in $Ln\ P_i$ due to the passage of time, holding all other variables, including quality, constant (Berndt, 1991). Thus, $P_{it}/P_{i,(t-1)} = \exp(\gamma_t - \gamma_{t-1})$. The implicit dollar prices of the continuous quality characteristics (potency and toxicity) are obtained from $\partial P/\partial X = \beta P X^{\frac{1}{2}-1}$. The implicit price of persistence, a dummy variable, is e^{γ^1}.

After the quality-adjusted price indices of the herbicides used in each of the major crops are calculated, it is necessary to aggregate them across crops. Many index number formulae have been proposed, depending on how the averages are calculated as well as the weighing method (Fisher, 1938). Diewert (1976, 1978) defines an index as superlative when it is exact for an aggregator (utility or production) function that "can provide a second-order approximation to an arbitrary twice differentiable linearly homogeneous function." In addition, Diewert (1978) shows that all superlative indices closely approximate each other, that the choice among them is usually immaterial, and that all superlative indices are consistent in aggregation. Among the superlative indices the most commonly used is the Tornqvist-Theil (TT) approximation of the continuous Divisia index, which is exact for the homogeneous translog aggregator function (Diewert). In addition, the TT index passes important tests such as the time reversal test and (approximately) the factor reversal test. We use the chained version of the Tornqvist-Theil approximation.

To calculate the quantity indices it is necessary first to obtain the herbicide expenditures ($V_t = \Sigma_i P_{it} Q_{it}$). Assuming that the aggregate price changes of the herbicides used in major crops are representative of the aggregate price changes of herbicides used in all U.S. agriculture, we can also calculate aggregate quantity indices by dividing total herbicide expenditures in U.S. agriculture by the price indices and normalize, so that the indices meet Fisher's weak factor reversal test; i.e. the product of the price times the quantity index yields the expenditure ratio between the two periods ($P \cong Q = V_t/V_0$) (Diewert, 1976).

4. ALTERNATIVE SPECIFICATIONS AND EXTENSIONS OF THE ECONOMIC THRESHOLD CONCEPT AND THE CONTROL OF LIVESTOCK PESTS

Rex Davis and Clement A. Tisdell

ABSTRACT

This paper outlines economic threshold models developed by various authors as an aid to decision-making about pest management and their applicability to pests of livestock. The definitional confusions relating to economic threshold models are raised as are limitations for applying threshold models. Complexities in the nature of yield loss function due to uncertainty in pest densities, the presence of multiple-pests, and the occurrence of pesticide resistance are discussed. An extension is provided that incorporates both multiple-pest species and pest resistance to control measures. Complications relating to the cost functions for pest control are considered. The combination of these factors limits the applicability of profit-maximising thresholds for livestock management, especially compared to other strategies such as prophylaxis.

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INTRODUCTION

The economic threshold is the most frequently applied technique in the field of economic pest management. The concept of linking the pest population to a treatment decision was first formalised by Stern et al. (1959). A key to the popularity of the original concept has been the combination of practicality and simplicity. This has made it the natural choice of applied entomologists and agronomists. The variety and quantum of economic threshold applications have been outlined by Peterson (1996). He finds that the vast majority (81.9%) of the reviewed scientific literature relates to insect pests, with the majority of these applications focused on cropping situations. Of the other applications by pest-types, weeds and plant diseases have also received considerable attention, again with a focus on crop protection.

The popularity of the concept of the economic threshold for pest control decisions has emerged despite divergent definitions. In particular, the work of Headley (1972), and subsequent modifications by Hall and Norgaard (1973) present a definition that is substantially different to the original concept defined by Stern et al. (1959). Interestingly, the concept of the economic threshold has never been as popular in livestock pest management as in crop management. Certainly research on the economics of managing the cattle tick *Boophilus microplus*, a major pest species in Australia, has focused on strategic (prophylaxis) treatments rather than identifying threshold levels. Nevertheless, Jonsson and Matschoss (1998) indicated that threshold-style decisions for treatments of cattle ticks are taken by 50% of dairy farmers in Queensland.

Given the complexities of modern agriculture, such as the presence of multiple-pest species and insect resistance, do threshold-based treatments provide satisfactory economic theories for pest management or are they simple, broad rules of thumb which represent the best alternative from a small list of alternative strategies?

This paper discusses the usefulness of the economic threshold concept particularly in relation to the modern management of pests of livestock. The paper analyses these issues by examining the scope of threshold treatment strategies amongst all pest management options. It then examines the definitional divergence and confusion in the economic threshold literature and the importance of specification, in both functional forms and in the variables included. Extensions to the economic threshold are considered as is an illustrative model that highlights the additional implications where fixed and application costs exist in the treatment of livestock. Conclusions and future research potential are then provided.

ECONOMICS OF PEST MANAGEMENT AND THE ECONOMIC THRESHOLD CONCEPT

Norgaard (1976) provides a base model of the economics of pest management. The term 'pest management' in this present context encompasses all actions undertaken by producers against pests. For an individual producer, the returns from conducting pest control are the increases in the net monetary value of yield resulting from the pest management technique. A monetary value for yield normally also involves issues about product quantity and quality. The total costs of a pest management strategy can include the costs of acquiring information, the costs of pest management inputs and the costs of applying those inputs. The economics of the firm state, *ceteris paribus*, that a producer will use a variable input up to the point where the marginal revenue product from that input is equal to the marginal cost of using that input. Fox and Weersink (1995) observed that inputs designed to prevent damage provide unique problems for economists because in contrast to conventional inputs, damage control inputs operate through an indirect effect on output. The choice of damage control inputs will depend upon the strategy used by the producer in a given period.

Cousens (1987) suggests that there are three distinct pest management strategies:

- *eradication* – this is a strategy in which extensive efforts and costs are provided in the short term to completely remove the pest and therefore provide unhindered produce development in future periods;
- *prophylaxis* – this is a strategy of insurance, in which pest controls are applied systematically, periodically and generally preventively regardless of the pest population;
- *containment* – the intention is to ensure the pest population stays below a specific level. The producer in this situation accepts some loss of yield (and therefore revenue) and controls the pest when it is cost-effective to do so.

Usually no single pest management strategy is dominant for any given pest. In cattle tick control in Queensland for example, a prophylactic or strategic dipping program has been shown in research trials in certain areas to be economically superior to a containment strategy (see Burns et al., 1977). However in other regions across the State, variance in the tick population may mean that cattle are only chemically treated when pest populations reach a certain threshold.

In many situations, technical constraints limit the number of alternative approaches available to a producer. For example, in the control of the cattle tick because of tick mobility and therefore externalities, eradication for an individual producer is unlikely to be successful without the assistance of neighbours and is disregarded as a viable pest management method (Cattle Tick Control Commission, 1973). The essence of economic pest management is to determine which pest management strategy class is viable or preferred, and then optimise the actions taken by using that strategy. For example, if a containment strategy is determined to be the only practical solution for pest management, then the role of economic pest management models is to determine what level of pest population should be tolerated and treated with what intensity. Generally this will involve calculating an economic threshold.

Two points can be derived from the above discussion. First, from the point of view of an economic analyst, it is vital to have appropriate methodologies to compare the relatives strengths and weaknesses between the different strategies, and then determine the optimal application within the preferred strategy. Secondly, the importance of economic threshold treatments are limited to a sub-class of pest management strategies, although containment is generally the dominant form of pest management strategy undertaken by agricultural producers.

Having considered the economic threshold in terms of its relationship within overarching pest management strategy alternatives, we now turn to a definition of the economic threshold itself. The definitional debate and confusion that has existed since the seminal work by Stern et al. (1959) is ironic given the conceptual simplicity of the original model. To calculate an economic threshold a practitioner needs to first estimate the economic injury level (EIL). The economic injury level is the pest population density that will result in economic damage. Stern et al. (1959) defined economic damage as the point at which the "amount of injury justifies the cost of artificial control measures". The economic threshold is the pest population density at which control measures should be adopted to prevent an increasing pest population reaching the EIL.

In essence, although the Stern et al. (1959) models are described as "economic threshold" models, the major component of economic calculations occurs in estimating the EIL. The economic threshold is simply the operational criteria for administering pest control action (Higley & Pedigo, 1996). The generalised form of the EIL described by Pedigo et al. (1986) is:

$$EIL = \frac{C}{VDIK} \tag{1}$$

where *EIL* is the economic injury level described in injury equivalents per production unit, such as insects/ha,

C is the management costs per production unit (\$/ha),
V is the market value per unit of production (\$/kg),
D is the damage per unit injury (kg reduction/ha/injury),
I is the injury per pest equivalent (injury/insect) and
K is the proportional reduction in injury with control.

The resulting measure from an EIL calculation will be a pest population which relates to the point at which the costs and benefits of control are equal.

There are three non-exclusive issues that have led to an array of subsequent extensions following the model outlined by Stern et al. (1959):

(1) *What constitutes "economic damage"*? Stern et al. (1959) presented a form of break-even analysis. Should a threshold model be viewed as a break-even analysis for a single input or should it be a profit-maximising input (marginal benefits equals marginal costs) as defined by Norgaard (1976) above?
(2) *What defines the point of action*? How crucial is the economic threshold, the point of action, to the economic quantification of the EIL? If the threshold is surpassed are the implications catastrophic or incremental? In other words, what functional form does the model of yield damage follow.
(3) *What other variables should be considered in threshold models*? If other variables are considered, such as a producer's attitude to risk, multiple-pest species, or chemical resistance, does the EIL or the economic threshold increase or decrease?

The first of these issues was initially raised by Headley (1972) who observed that the Stern et al. (1959) break-even definition of "economic damage" was deficient as a profit-maximising model. The alternative model developed by Headley (1972), in its most basic form, consisted of three variables. These being damage to the product caused by the pest, the pest population and time. The economic threshold as defined by Headley (1972a) is described graphically in Fig. 1. The upper half of the diagram depicts the overall cost of pest control and the value of production at each pest population level. The value of production is highest with a pest-free environment and then remains constant until the pest population reaches a critical mass of which it then begins to reduce the yield of the product. The cost of control is highest when the population is kept to zero and follows entomological observations that the cost of reducing a pest population increases substantially if attempts are made to achieve very high kill rates (Headley, 1972).

The lower half of the diagram displays the marginal values of production and control costs. The economic threshold (according to Headley's (1972)

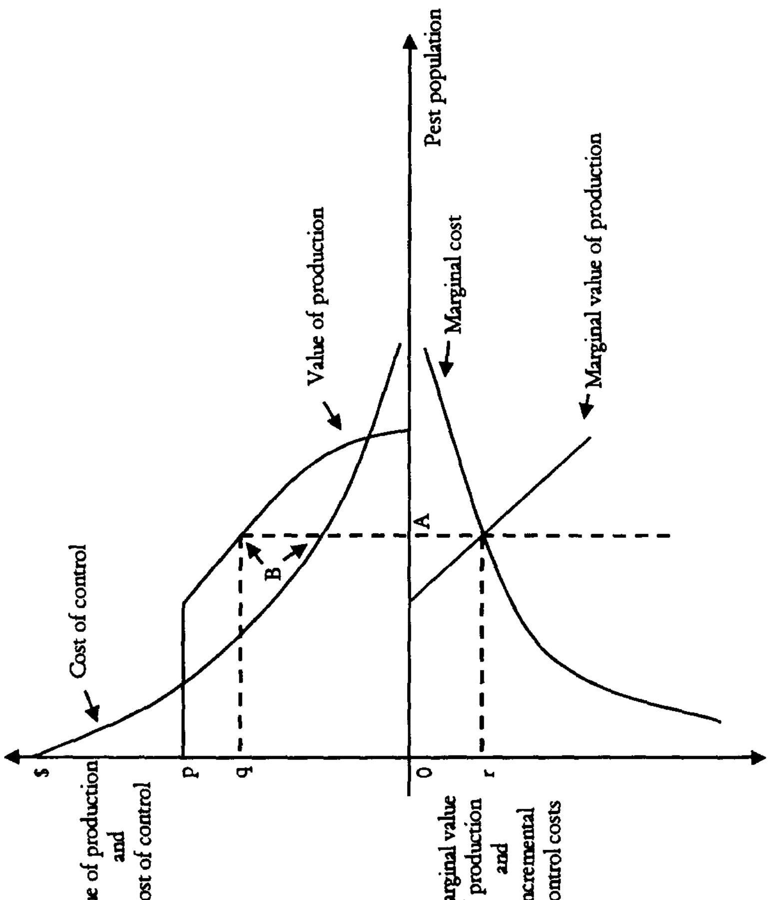

Fig. 1. The Economic Threshold for Pest Control.

Source: Based on Headley (1972, p. 103).

viewpoint), corresponds to the pest population at which the marginal value of production is equal to the marginal cost of pest control. In this case, this implies a value of r for these functions. In Fig. 1, the corresponding economically optimal pest population is $0A$. In this model, it is better to sacrifice the value of production between p and q then to try to maintain yield at level p as the costs of reducing the pest population to this point expands exponentially.

Pedigo (1996) observed that when Headley described the economic threshold he in fact was describing the EIL. Hall and Norgaard (1973) observed that Headley's definition of the economic threshold level actually represents a post treatment population level rather than a trigger for action as defined by Stern et al. (1959). Hall and Norgaard (1973) corrected the economic threshold to a pre-treatment population level and include timing of the treatment variable and the dosage of the control technique flexible and inherent components of the model. The modifications made by Hall and Norgaard (1973) could be interpreted as being an all encompassing threshold definition in which Stern et al. (1959) model is a sub-set in which the costs are fixed or linear.

The modification by Headley, Hall and Norgaard, however, has not been viewed in the literature as relatively aligned approaches but rather have led to a dichotomy in relation to economic thresholds, between approaches that establish EIL based on a profit-maximising model and those of the Stern et al. (1959) tradition based on break-even analysis. This point that has been observed by Hall and Moffitt (1985), Moffitt (1986), Plant (1986), Weersink et al. (1991), and Higley and Pedigo (1996).

Despite the theoretical strengths of the Headley-Hall-Norgaard concept in terms of its linkages to economic principle, the weight of literature has favoured the initial Stern et al. (1959) approach (Peterson, 1996). The dominance of the Stern et al. (1959) approach has been explained by Moffitt (1986), Plant (1986) and Weersink et al. (1991), who state that the definition simply highlights differences in attitudes between entomologists and economists. This appears unsatisfactory as many economists also continue to define economic thresholds in terms of the Stern et al. (1959) concept, for example Auld and Tisdell (1987).

A possible reason for the popularity of break-even thresholds is that they provide simple and effective decision-rules that can be empirically derived and applied by practitioners in the field. On the other hand, the degree of knowledge required to establish an economic threshold suggested by Headley-Hall-Norgaard is considerable. For that threshold to be applied, information on the whole cost function is required as opposed to a discrete point for the Stern et al. (1959) threshold. Furthermore, in many situations dosage rates are prescribed thereby limiting the evaluation to a break-even analysis. While this may not maximise profit, it at least ensures that a control measure can be justified on some economic grounds.

Hall (1988) accepts the criticism of complexity and extends the argument by stating that the prescriptive value of the economic threshold models described above and many of the applications of the economic threshold are limited due to the specification and experiments on which the models are based being specific to individual situations. As Hall (1988, p. 642) states: "It is difficult enough for Ph.D. agricultural economists and entomologists to develop these models, design experiments and estimate parameters, much less expect that each farmer will do so . . ."

A third and less discussed possibility is that thresholds are based on producer behaviour other than profit maximisation and are more related to a producer maximising expected utility. In these situations the producers attitude to risk becomes important.

According to Pannell (1990), several stochastic variables are likely to be observed in any economic pest management model. First, uncertainty can occur due to a lack of knowledge of the initial pest density or a lack of certainty in relation to the number of pests killed. Second, uncertainty can be attributed to a lack of knowledge of the pest-free yield as well as a limited understanding of the actual damage function as mentioned above. Uncertainty, therefore, has a direct and often major effect on profit. Furthermore, pesticides involve a form of insurance against pest damage and therefore a potential reduction of risk (Norgaard, 1976).

If uncertainty is present, attitudes of producers to risk need to be examined. Feder (1979) developed a comprehensive utility model that examined management techniques based on producers' risk profiles and finds that unlike other industries in which the presence of risk leads to a decrease in inputs, uncertainty is likely to increase pesticide use. Moffitt (1986) on the other hand examines risk based on Stern et al. (1959) economic thresholds. In his model, producers do not necessarily increase their inputs when considered over the course of a season, rather risk aversion will manifest itself in higher pesticide dosage. Tisdell (1986) and Auld and Tisdell (1987) indicate that there are a considerable variety of producer responses to pest management when uncertainty is present particularly when assumptions of risk aversion are relaxed and replaced with risk neutrality or a risk preference.

Plant (1986) and Szmedra et al. (1990) are highly critical of the use of economic thresholds in the presence of uncertainty. Plant (1986) finds that the critical value of pesticide dosage (economic threshold) increases with increasing uncertainty. However, as opposed to Feder (1979) this is claimed not to be due to risk-aversion but because the expected mortality rate of pests decreases with higher levels of uncertainty. That is, the level of variance in the model is reduced with increased pesticide dosages as nearly all the pests are killed. Plant (1986) questions the use of economic thresholds at all because inclusion of additional levels of uncertainty and taking into account the natural dynamics of pest control mean that techniques such as sequential decision theory are better equipped to provide pest management advice.

Cousens (1987) identified a number of additional types of threshold in relation to uncertainty and producer risk profiles. These include safety thresholds, which refer to producers tolerating lower pest population levels or damage when applying treatments due to their aversion to risk. Similarly visual thresholds refer to the fact that many producers will make their decisions on their own perceptions of the pest population regardless of scientific or extension advise.

The points made by Plant (1986) and the safety threshold identified by Cousens (1987) are particularly important. The implications of higher levels of risk aversion or satisficing behaviour in producers is that threshold models began to resemble a strategy of prophylaxis. That is, if a producer is aiming for a minimum outcome rather than a profit-maximising approach, pest treatments are more likely to based on calendar dates rather than with reference to the pest population.

Aside from the issues in relation to producer attitudes to risk, specification is important in terms of establishing both the EIL and economic threshold. For example, the models above have focused on the main cost of pest management being chemical control. In livestock issues one of the main costs is application. Mustering of the cattle, particularly on cattle stations with low cattle density provides the majority of costs.

Fox and Weersink (1995) observe that many damage functional forms can arise. Although conventional wisdom is to examine relationships that result in decreasing returns to pest management, situations may exist where increasing returns from the damage control input are possible depending on the model specification. They observe that increasing returns highlight the potential for corner point solutions such as that provided above.

Specification becomes more important in relation to the economic threshold. So far this paper has not discussed the importance of the difference between when the pest population is treated to avoid reaching the EIL. Depending upon complex issues of pest dynamics, the timing of the treatment may be crucial or unimportant.

In addition, Cousens (1987) identifies a threshold which he calls the competition threshold. The competition threshold arises due to the possibility that a sigmoidal-like relationship can be observed between weed density (his field of interest) and yield as opposed to the normally observed hyperbolic yield – weed relationship. When a sigmoidal relationship is observed This differs from the classic hyperbolic yield function in which damage begins instantly and increases at a decreasing rate until only a limited level of produce is left to save. In a sigmoidal relationship, a period exists in which no damage is recorded until the weed density reaches a certain critical mass at which the level of damage increases significantly. The point at which damage begins in the sigmoidal response function is the competition threshold – named as the point at which weed density competition begins to effect yield. The important element from Cousens (1987) identification of competition thresholds is that it highlights the fact that yield response relationships can give rise to a range of critical density points that may be considered as catalysts for action.[1]

Apart from Cousens (1987), Pedigo et al. (1986) also provides a taxonomy of different threshold approaches based on different definitions of functional form and yield response curves. While Cousens (1987) classifies an array of new threshold terms, Pedigo and Higley (1996) claim that the main purpose of the Pedigo et al. (1986) is to discourage further new terminology and the misuse of the original economic threshold concept.

Figure 2 displays two conceptual yield response curves. These highlight the importance of identifying response relationships when specifying thresholds and follow similar relationships to those originally observed by Pedigo et al. (1986).

In (*a*) a situation is identified in which past a certain pest density all produce is lost, accentuating the importance of treating before a particular pest density. In (*b*) a situation is identified where at low levels the "pest" actually increases yield at low levels but beyond a certain point becomes unmanageable and decreases yield. While these examples are conceptual and extreme, they do raise interesting possibilities and may have some grounding in producer behaviour. If a pest is able to inflect mortalities once a certain density is reached then situation (*a*) would apply. Situation (*b*) could relate to evidence from some cattle producers in Queensland who indicate that a small tick presence ensures the maintenance of high levels of immunity to tick fever.

The difference in functional form relating to the damage function highlights one of the real difficulties in applying economic thresholds.[2] For an application to be successful, the agronomist or other such practitioner needs to have at least some knowledge of the relationship between the pest and yield. In many cases, this condition will not be fulfilled making some thresholds applications as inappropriate. Campbell and Thomas (1996) observe one of the reasons why livestock economic threshold applications have been limited in their application to veterinary pests is that the damage function is likely to be both more

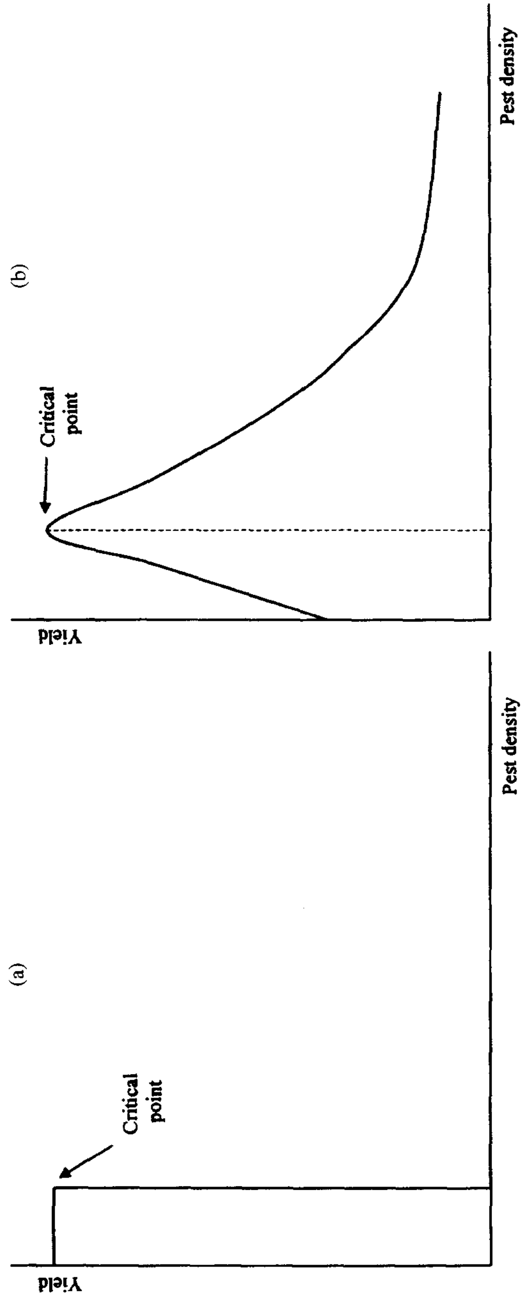

Fig. 2. Other Conceptual Response Curve Examples.

complicated and more subtle than that observed in cropping as pests of livestock are often vectors for disease (such as ticks and the transmission of tick fever) and this relationship is difficult to quantify.

EXTENSIONS OF THRESHOLD MODELS

Aside from uncertainty mentioned above, other important variables often need to be incorporated into producer decision models about pest control especially:

(1) **Environmental variables** – in situations where there is a divergence between private and social cost functions, the inclusion of additional variables to correct for externalities such as environmental damage may be appropriate.
(2) **Chemical resistance** – each year the level of insect resistance to chemical control measures continues to rise. Part of this rise is due to inappropriate chemical control measures which intensify the problem.
(3) **Multiple-pest species** – traditional economic thresholds only examine one pest species at a time. Treatments can be based not only on achieving control in a primary pest species but also in ensuring a reduced pest population in another pest species. The relationship treatment and the other pest species is often not considered.

The concept of an environmental economic threshold has been developed by Higley and Wintersteen (1992). They use a contingent valuation approach to identify an environmental cost factor which is incorporated into a standard EIL equation. This relatively simple but effective extension provides a mechanism for establishing the impact on individual producer pest management decisions when environmental or health hazards are important.

Several papers have examined the role of pesticide resistance. Tisdell (1982) in a generalised framework observes that when the effectiveness of techniques decline over time, and the effect of this loss is known, then welfare maximisation over multiple time-frames may be maximised by reduced consumption of the technique in the current time period.

Specific resistance models have been developed by Hueth and Regev (1974) and Taylor and Headley (1975). In these models Hueth and Regev (1974) observe that the economic threshold not only changes between seasons, due to changing effectiveness of the pesticides, but within a particular season as well. They find that the economic threshold increases over a season as a producer is willing to forego more yield the closer the product is to harvesting. More importantly, the also find that the exclusion of the variable relating to increase resistance to control, only results in an overuse of chemicals with additional restrictive assumptions. In other words, as timing and dosage are able to be varied in their model, the actual effect on chemical use is unclear due to the varying incremental effects of the effectiveness of pest control applications across a season.

Taylor and Headley (1975) find that the use of pest population functions which incorporate resistance will result in an improved pest control decision, provided that the additional benefit of this decision (the benefits gained from making the greatest use of a control technique over time) is greater than the cost of acquiring the information necessary to provide greater pest population modelling.

The existence of multiple-pest species is an issue which has dogged the use of economic thresholds. Apart from difficulties in defining the damage function, Campbell and Thomas (1996) highlight multiple-pest species as a further factor behind the lack of economic threshold applications to veterinary pests.

Multiple-pest models have been developed by several authors. These models can be divided into two sub-classes: whether the pests are independent of each other in terms of their consequences on yield; and whether all the "pests" have negative implications for yield. Palis et al. (1990) provides an example of a model in which two pests exist and both have negative implications for yield. They use iso-loss lines to determine multiple-pest species economic thresholds. Iso-loss lines indicate combinations of the pest species that result in the same loss of yield. When a combined pest population exceeds the iso-loss line, then treatment is justified.

Swinton et al. (1994) examined situations of multiple species of weed by modifying the hyperbolic yield function identified by Cousens (1987) to create a nonlinear competitive index of interactions between weeds and crops. Their model developed statistical relationships between eight weed classes and their impacts between two crop types (corn and soybean) for 13 locations. This method resulted in the establishment of competition coefficients which indicate the percentage of yield lost per weed. The Swinton et al. (1994) model is particularly useful given sufficient data, as it can also be used to examine the second class of multiple-pest situations in which chemical control of one pest, can damage other neutral or friendly species. For example, a chemical control may kill natural predators of a pest species, which may lead to a secondary pest outbreak, or have other similar consequences that reduce the yield. A version of this problem was examined by Auld et al. (1987) and is discussed graphically using Fig. 3.

In Fig. 3, a series of conceptual net profit functions are presented. Profit functions 1 and 4 are classical situations in which increased pest management intensity brings increasing benefits to a maximum (such as the low rate of control, P_1 for profit function 1 or a high rate of control P_2 for profit function 4), and then the benefits decrease with increased pest management intensity as the proportion of additional yield protected declines. Complicating factors, such as natural predators of the pest, which are also susceptible to control mechanisms, indicate that other possibilities such as profit functions 2 or 3 may exist. In these situations, low levels of pest management intensity do not damage the predator of the secondary pest species. However, increasing pest management intensity, aimed at the primary pest, gradually begins to impact on the predator of the secondary pest past P_1 and its reduction results in a secondary pest outbreak that reduces yield. Given profit function 3, this

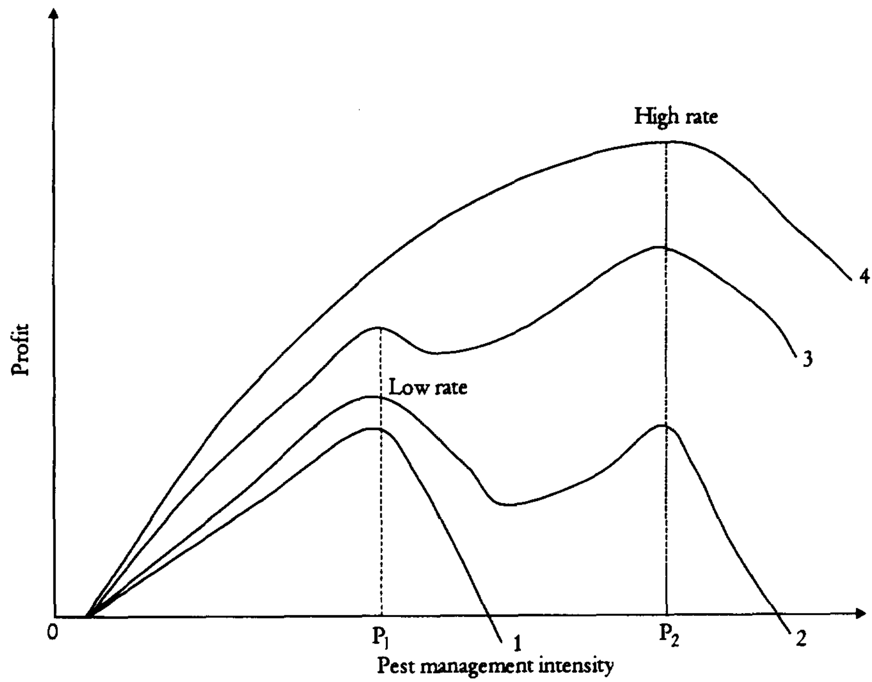

Fig. 3. Possible Pest Functions for Pest Management where a Non-target and Predatory Pest Species is Effected by Increasing Pest Management Intensity.

Source: Adapted from Auld et al. (1987).

reduced yield may be rectified and profit increased by intensifying the level of pest management i.e. to ensure treatment impacts on the secondary pest as well. By contrast, the increased pest management intensity for the secondary pest outbreak if profit function 2 applies, results in a maximum profit less than that attainable before the secondary pest outbreak is triggered.

Profit functions 2 and 3 highlight another situation involving multiple maxima. The humped profit curves, indicate a low local profit maximum at P_1, and a high maximum at P_2. The issues involving multiple maxima can be considered in more detail by use of Fig. 4. In this example, a marginal cost function, MC_1 represents the marginal cost from controlling the primary pest and increases exponentially as it is becomes more difficult to provide 100% protection.[3] The marginal benefit curve, MB_1, has two humps, with marginal benefits initially increasing with greater protection of the crop from the primary pest, and then decreasing as the natural predator of the secondary pest is destroyed and a secondary pest outbreak occurs. The curve then increases as the higher pest management intensity results in protection from the secondary pest also, and then decreases to zero as 100% crop protection is provided and no additional benefit from pest management is possible.

Consider the marginal cost curve marked MC_1. The intersection of MC_1 with MB_1 occurs at a local profit minimum corresponding to B and the optimum level of pest management intensity is P_6 corresponding to a local profit maximum at G. This local profit maximum is also a global one. In this situation, the producer would operate at a high level of pest management intensity and severely damage the predator of the secondary pest species. If, on the other hand, the marginal cost curve of control marked MC_2 applies, there are two minima at C and E (with associated levels of pest management of P_2 and P_4) and two local maxima at D and F. The profit-maximising level of pest management can then only be determined by examining the difference in total profit between D and F. It is possible that profit could be maximised in such a case by a 'low' level of control of the primary pest, a level corresponding to point d, because this low level of control is less favourable to proliferation of the secondary pest.

Szmedra et al. (1988) uses a simulation model to examine the interactions of two-pest species in a pest decision framework. However in their simulation model natural predators of certain pest species are also killed by the main pesticide leading to additional outbreaks. Harper and Zilberman (1989) provide a model that examines secondary pest outbreaks caused by chemical treatments killing not only the primary pest but predators of the secondary pest. In none of these models are the implications of multiple-pest species and chemical resistance examined together.

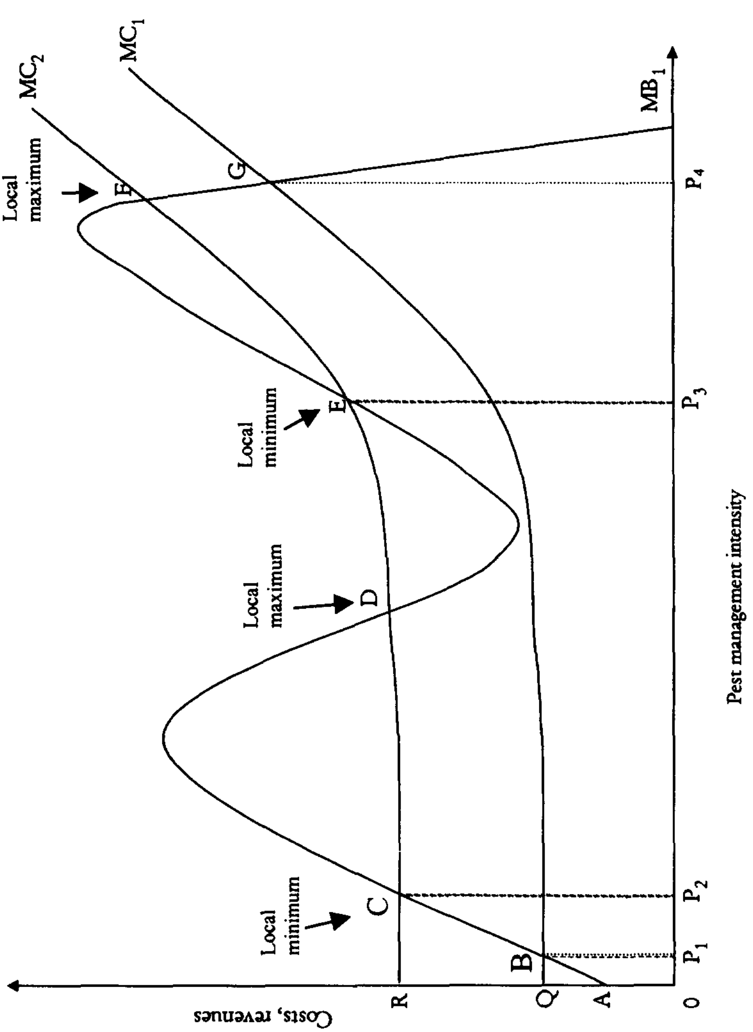

Fig. 4. Marginal Analysis for an Individual Producer Faced with Two Pest Species where a Predator of the Secondary Pest Species is Affected by Higher Levels of Pest Management Intensity, or where the Secondary Pest is Otherwise Favoured by Control of Primary Pest e.g. by Less Interspecies Competition.

IMPLICATIONS OF THE THRESHOLD CONCEPT WITH MULTIPLE PESTS, CHEMICAL RESISTANCE AND HIGH FIXED AND APPLICATION COSTS

The framework present by Harper and Zilberman (1989) is extended to examine potential resistance implications for producers stemming from multiple-pest species management decisions. The Harper and Zilberman (1989) approach has been selected as it is a production theory approach which examines damage as a proportion of potential yield, and allows the incorporation of pest populations into the damage function. An important element of the model presented below is to examine the scenarios arising from different cost structures.

The initial assumptions of the model are that there is an agricultural producer whose product is attacked by two different species of pest. At this stage there is no assumption as to which species is the predominant pest. It is also assumed that the main form of pest-control is through pesticide applications.

The grazier's production function is equal to:

$$Q = f(X)[1 - D\{S_1, S_2\}], \tag{2}$$

where Q is equal to quantity, X is the non-pesticide input and $f(X)$ is the potential output without any damage from pests with $f' > 0, f''' < 0$, $D\{S_1, S_2\}$ is the damage function where S_1 represents the population of pest species 1 and S_2 represents the population of pest species 2.

The damage function D expresses the fraction of yield lost because of both pests. It is assumed that damage is directly related to the size of the population and expresses the yield lost because of both pests. No natural predators for either species are considered.

$$D = D\{S_1, S_2\} \tag{3}$$

where D_{S1}, $D_{S2} > 0$. The population equations for the two pest species are:

$$S_1 = k_1(X)[1 - M_{1i}(Z_i)]R_{1i} \tag{4}$$

$$S_2 = k_2(X)[1 - M_{2i}(Z_i)]R_{2i} \tag{5}$$

where k_i is the carrying capacity that would be achieved by the insect population if no pesticide is used, M_{1i} is the mortality rate caused by the dosage of pesticide i for species 1, M_{2i} is the mortality rate caused by the dosage of pesticide i for species 2, Z_i is the dosage of pesticide i, R_{1i} is a measure of pesticide resistance by species 1 to pesticide i where $R_{1i} > 1$, and R_{2i} is a measure of pesticide resistance by species 2 to pesticide i where $R_{2i} > 1$.

The purpose of the variables R_{1i} and R_{2i} is to offset the decreases in population from the term $[1 - M_i(Z_i)]$. For example, if the mortality rate for $M_1(Z_1)$ is 0.9 then $1 \leq R_{11} \leq 10$.

The model therefore states that a producer's production function will be determined by the potential yield, which is dependent on the non-pesticide inputs into the production process, and the fraction of the crop that is lost in damage to the two pest species. The amount of damage is determined by the population equations for the two pest species which are in turn determined by the carrying capacity achieved due to the non-pesticide production input, the mortality rates of the pest species resulting from pesticide applications, and the subsequent level of resistance to the pesticide used.

It is also assumed that the producer has a choice of three chemicals, Z_1 represents the quantity of a pesticide that is used to control pest species 1 but this pesticide has a negligible effect on pest species 2. Z_2 is the quantity of a pesticide that is used to control pest species 2 but this pesticide has a negligible effect on pest species 1, and Z_3 is the quantity of the pesticide that can control both pest species. The producer's cost function is equal to

$$C = uX + a_1 + a_2 + a_3 + w_1Z_1 + w_2Z_2 + w_3Z_3 + y_1 + y_2 + y_3 \tag{6}$$

where u is the cost of the non-pesticide input, a_i are the fixed costs associated with applying pesticide i, w_i is the cost of pesticide i and y_i is the cost of applying pesticide i.

If π is profit, and p is the price received for the producer will aim to maximise profits subject to the pest population levels, so that

$$\max \pi = pf(X)[1 - D\{S_1, S_2\}] \quad uX - a_1 - a_2 - a_3 - w_1Z_1 - w_2Z_2 - y_1 - y_2 - y_3 \tag{7}$$

subject to

$$S_1 = k_1(X)[1 - M_{1i}(Z_i)R_{1i} \tag{8}$$

$$S_2 = k_2(X)[1 - M_{2i}(Z_i)R_{2i} \tag{9}$$

To examine the possible implications of this model a number of situations are examined. In the first instance let us assume that the fixed, application and unit costs of the three pesticide chemicals are the same, that is, $w_1Z_1 = w_2Z_2 = w_3Z_3$, $a_1 = a_2 = a_3$, and $y_1 = y_2 = y_3$. Let us also assume that pesticide 3 has the same effect over S_1 as does pesticide 1, and that pesticide 3 has the same effect over S_2 as pesticide 2, $M_{11} = M_{13}$ and that $M_{21} = M_{23}$. Finally, it is also assumed that there is no resistance, $R_{ii} = 1$ and does not increase over time.

In this situation the producer will always choose pesticide 3 in every circumstance. If both pests required control in their own right, then pesticide 3 produces the saving of the fixed, application and unit costs of a second application of chemicals. Even if there is only one pest causing significant damage to the product, the choice of pesticide 3 still brings about more benefits through its ability to reduce the population of the second pest species. However, with the incorporation of different mortality rates, different cost structures and chemical resistance the choice is less obvious. To show the possibilities that emerge when these factors are considered two examples are considered. The first example examines the situation of a primary economically significant pest species that requires treatment in its own right, while the second situation example examines the situation where the cumulative damage function of both species requires treatment, however, control of a pest species when examined in isolation is not justified.

Example 1

In the first situation it is assumed that pest species 1 is the primary pest, that is $D_{S1} > D_{S2}$ and that the damage inflicted by Species 1 on the product is sufficient enough to warrant its control, so that:

$$pf(X) - pf(x)[1 - D\{S_1\}] > a_i + y_i + w_i Z_i \qquad (10)$$

where $i = 1$ or 3. It is also assumed that although Species 2 is damaging the product, its population level does not justify control in its own right:[4]

$$pf(X) - pf(x)[1 - D\{S_2\}] < a_2 + y_2 + w_2 Z_2 \qquad (11)$$

In this situation the producer has two options which are detailed in Table 1.

Table 1. Control Options for a Producer where Damage is Accumulating Through 2 Pest Species with Only 1 Justifying Control in its Own Right.

Option A	***Option B***
Apply pesticide 1 which controls just species 1. The producer's cost function is:	Apply pesticide 3 which controls both pest species. The producer's cost function becomes:
$C = uX + a_1 + y_1 + w_1 Z_1$	$C = uX + a_3 + y_3 + w_3 Z_3$
with the population equations for the initial time period:	with the population equations for the initial time period:
$S_1 = k_1(X)[1 - M_{11}(Z_1)]R_{11}$	$S_1 = k_1(X)[1 - M_{13}(Z_3)]R_{13}$
$S_2 = k_2(X)[1 - M_{21}(Z_1)]R_{21}$	$S_2 = k_2(X)[1 - M_{23}(Z_3)]R_{23}$
where $0 < M_{11}(Z_1) < 1$, $M_{21}(Z_1) = 0$, $R_{11} = 1$ and $R_{21} = 1$	where $0 < M_{13}(Z_3) < 1$, $0 < M_{23}(Z_3) < 1$, $R_{13} = 1$ and $R_{23} = 1$

To concentrate on the effect of resistance, the assumption that $M_{11} = M_{13}$ is retained as is $w_1 Z_1 = w_3 Z_3$, $a_1 = a_3$, and $y_1 = y_3$ so that there is no cost advantage involved for either pesticide. Let us also assume that resistance in the primary pest S_1 is negligible, however, $R_{21}^{t+1} > 1$. In this situation, the producer has to determine whether the present value of benefits from controlling S_2 justify the decreased effectiveness of the technique at a later date. This situation is made more interesting if $M_{23}(Z_3) > M_{22}(Z_2)$ and that $a_3 + y_3 + w_3 Z_3 < a_2 + y_2 + w_2 Z_2$. In this situation, increased resistance to pesticide 3 by S_2 has a much higher cost, as pesticide 3 is the most effective and less expensive form of control against S_2. In this circumstances, the producer may decide to choose pesticide 1 and this is even more likely if $a_1 + y_1 + w_1 Z_1 < a_3 + y_3 + w_3 Z_3$.

Example 2

In this situation it is assumed that:

$$pf(X) - pf(x)[1 - D\{S_1\}] < a_1 + y_1 + w_1 Z_1 \tag{12}$$

$$pf(X) - pf(x)[1 - D\{S_2\}] < a_2 + y_2 + w_2 Z_2 \tag{13}$$

However:

$$pf(X) - pf(x)[1 - D\{S_2\}] < a_3 + y_3 + w_3 Z_3 \tag{14}$$

If the remaining assumptions utilised at the beginning of Example 1 are retained, then for this situation producer is more likely to trade-off future resistance to chemical control of pesticide 3 for the extra benefits of pest control in this current season. The producer also knows that pesticides 1 and 2 are available if required at a future date if pesticide 3 proves ineffective in the long run. As in the last example, adjusting the costs of application, the relative mortality rates of the pesticides, and the rate of resistance may provide different outcomes.

IMPLICATIONS FOR THE IMPLEMENTATION OF THE ECONOMIC THRESHOLD FOR PESTS OF LIVESTOCK

The illustrative model above simply highlights situations in which complications such as multiple-pests, chemical resistance and cost structures have an impact on the pest management decision. Further development of the above examples would lead to an incremental analysis which would determine the points at which the marginal benefits of treating species 1, species 2 or both species, would equate to the marginal cost of using pesticide 1, pesticide 2 or pesticide 3 plus the cost of the declined effectiveness of the control technique in future seasons.

What is clear however is that the role of fixed cost can have a major impact on pest control decisions. In this case, the choice of pesticide 3 is more appealing in many cases as the fixed costs and application costs in providing two separate treatments for each pest may be substantially larger than that for one.

While the above model is general, it is relevant to the application of economic threshold decision techniques in the management of pests of livestock. As mentioned earlier in the paper, there has been limited application of economic threshold approaches to livestock. This may be because livestock situations combine all of the complexities and extensions impacting on pest management decisions mentioned above. Livestock pest management often involves treatment of multiple pests, there are environmental and human health risks from chemical contamination of meat and milk, and pests become resistant to chemical control measures.

However, additional complexities arise from the fact that livestock are mobile, grow or survive over multiple seasons, involve non-linear consequences from the outbreak of pests and diseases, and suffer quality as well as quantity impacts from pest populations. These add to the difficulties of calculating EILs for pests of livestock.

The illustrative model in the previous section, provides a small but highly applicable example of the added complication of cost functions in livestock situations. Here, the cost of mustering cattle and treatment facilities becomes the greatest expense for cattle producers conducting treatments. In these situations, the choice of dosage, as discussed in such detail in economic threshold models above, few producers having to outlay hundreds of dollars to muster cattle are going to trade off lower pesticide application dosages and pesticide efficacy for the sake of small sums of money. In other words, when chemical costs are a minor component of the overall pesticide cost function, *ceteris paribus*, it is unlikely that producers would choose anything other than the recommended dose.

A further component to be considered is the role of other farm management practices on producer pest control decisions. Again using the example of livestock, other management practices can be utilised jointly with the control technique as was discussed earlier. The way in which costs are allocated with the existence of joint or common costs[5] in the production process will therefore have an effect on the pest management decision as indicated in Figure 4. The level of difference will depend upon the means by which costs are allocated across joint production processes (see for example, Billera et al. (1981) and Gal-Or (1993)). The existence of joint production costs are a major determinant in the establishment of economies of scope which result from the ability of a firm to produce two products in combination than it is for them to be produced individually (Panzar & Willig, 1981). In pest control situations however, only one end product is usually being developed, such as meat, however economies of scope apply to the production of goods that are inputs into the production of the agricultural product.

The above discussion indicates that there is another complexity that needs to be considered in economic threshold decisions. Aside from the complexity in determining pest dynamics, which is the focus of most economic threshold applications[6] there is potentially an equally complex procedure involved in determining the cost function particularly in relation to livestock pest management.

CONCLUSIONS

This paper has examined the role of economic thresholds both generally and specifically in the management of pests of livestock. Where thresholds have been established they have been the break-even methods identified by Stern et al. (1959) rather than profit maximising applications in the tradition of Hall-Headley-Norgaard.

Economic thresholds have had limited use in livestock pest management. The question that this paper has asked is whether limited applications for livestock are a peculiarity and there a gap in the literature needs to be filled, or alternatively, that economic thresholds are simply not useful in terms of livestock pest management.

The answer to this question is that while, theoretically economic thresholds have much to offer, there are limited situations in which they will offer producers much assistance. First, economic thresholds are a form of containment strategy, which is one of three overall strategies that a producer may adopt. Second, as issues such as uncertainty and a producer's risk profile are considered the current and future level of the pest population becomes less relevant and a producer's strategy merges towards one of prophylaxis.

Second, the complexity in establishing the correct specification of the threshold model, both in terms of the form of the damage and yield functions is considerable, especially for a profit-maximizing function.[7] This complexity escalates in the presence of complicating factors such as multiple-pest species and chemical resistance to the extent that an 'optimal' analytical solution is virtually impossible. Moreover, the nature of the cost function is very important in livestock husbandry as there are substantial non-chemical costs that have to be considered in any pest management decisions.

These issues highlight the potential for major prescriptive discrepancies between economic threshold models. Again, this leads to the possibility that strategic treatments, based on calendar dates, or other decision techniques based on the pest population are potentially superior in situations of inconsistent and occasional pest populations simply due to the lack of acceptable threshold advice. This is also likely in situations with predictable and constant pest problems showing minimal seasonal variation.

Nevertheless, thresholds are still of considerable economic importance to producers affected by intermittent pests or diseases, particularly when there is a major impact on yield if pests are untreated. In these situations, an increased emphasis on the role of cost functions, and the use of bio-economic simulation models which explore the impacts of chemical resistance and inter-pest species relationships have much to offer in terms of improved EIL and economic threshold prescriptions.

NOTES

1. Cousens (1987) observes that models that do not include sigmoidal relationships have potential areas of application. He therefore identifies a category of thresholds known as "statistical thresholds" which are the points observed through research experiments and simulation which potentially do not correspond with thresholds observed in the field.
2. Relationships such as those described in Fig. 2 can be estimated using statistical techniques described by Hudson (1966). Where sharp turning points occur, Hudson (1966) suggests the value of examining the function as a series of sub-models.
3. The original example from which this analysis was inspired was discussed in terms of a reduction in yield through damaging a product that could potentially increase yield in this season or future seasons. As the area of interest of Auld et al. (1987) were weeds, their example was discussed in relation to under sown legumes that could be damaged and would reduce yield. In livestock, an example is that increased pest management intensity may result in some sickness in the animal or indeed secondary pest outbreaks through damage to a primary or secondary pest predatory.
4. It is also assumed that the producer from time to time has circumstances whereby S_2 becomes the primary pest.
5. Joint costs are those that are expended in the production of two or more goods but cannot be separated. Common costs are used in the production of both commodities but are able to be used in separate proportions for the production of each good (Billera et al., 1981).
6. See Peterson (1996) for a comprehensive review of economic threshold applications.
7. Note that another type of threshold can arise from the nature of the objective function. For example, 'satisficing' (Simon, 1957), rather than profit-maximizing behaviour, might be adopted by a livestock manager. This case is not analyzed here.

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5. THE INFLUENCE OF PEST MANAGEMENT ADVICE ON PESTICIDE USE IN CALIFORNIA TOMATOES

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ABSTRACT

This paper uses survey data collected from tomato growers in California to determine the factors that influence pest control advisor participation and pesticide use recommendations. We find that advisor recommendations are dependent on the probability of infestation conditional on the calendar and on advisors' perceptions of growers' knowledge, while growers depend more on information obtained from observing their crop. We also determine that the pesticide use recommendations of advisors are, on average, higher than those of growers. Results demonstrate the incentives of grower and advisor pesticide use decisions that must be taken into consideration when regulatory policy is designed.

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INTRODUCTION

When production processes are subject to random effects, producers may get involved in monitoring activities in order to better modify their responses to the actual state of nature. Monitoring and scouting of fields reduce some of the uncertainty involved in pest management and are important elements in Integrated Pest Management (IPM) strategies. Successful IPM programs use a combination of crop and ecosystem information, bio-rational products, and chemical pesticides to protect crops from damage and minimize the impact of pest management on the environment. The use of chemical pesticides in an IPM program is influenced by the amount of effort devoted to monitoring, which is in fact the process of information gathering. The increased monitoring of crop damage and pest populations provides growers with information which allows for more direct and efficient use of pesticides where they are needed.

The amount and type of pesticides used on crops is influenced by more than just growers' actions (Zilberman et al., 1994; Campbell, 1993; Wolf, 1998). It is typically assumed that growers make the majority of decisions regarding pesticide use, but in an increasingly complex regulatory and marketing system, there are many agents involved in determining pest management practices. Some of the other factors which influence pest management decisions are: pesticide use regulations; pesticide requirements imposed by the financial institutions which lend money to agricultural operations and the retailers who enter into marketing contracts with growers; and the information and advice provided by university extension agents and pest control advisors. Understanding the influence of pesticide use advice is increasingly important as pesticide products and regulations becomes more complex and consequently farmers become more dependent on the specialized knowledge of pest control advisors.

This paper develops a simple conceptual framework to assess the factors that influence pest monitoring activities by growers and pest control advisors and then applies this framework to data sampled from tomato growers in California. In particular, we seek to understand what determines the extent of growers' reliance on pest control advisors' opinions, what determines the level of pest monitoring undertaken by individual growers, and whether or not the reliance on advisors' opinions increases or decreases pesticide use levels.

Past studies have empirically examined the influence of advisor recommendations and information on pesticide use levels. Results from a study by Feder (1979) suggest that employing pest management consultants leads to a reduction in overall chemical pesticide use. A study by Pingali and Carlson (1985) shows that the level of fungicide and insecticide use is dependent on growers' knowledge as measured by schooling and farming experience (Pingali & Carlson, 1985).

In another study, Carlson (1980) estimates growers' demand for pest management advice as a function of the level of available public pest information. Results from his study indicate that publicly available pest information can be substituted for private pest control advisor information depending on the relative costs of the two types of information (Carlson, 1980). A study undertaken by Moffitt et al. (1986) develops a model which estimates the implications of public pest infestation forecasts and private pest information on insecticide use and pest monitoring (Moffitt et al., 1986). Results of their analysis demonstrate that publicly available information forecasts leads to a reduction in the demand for consultants but to an increase in overall pesticide use if those forecasts are considered to be of low reliability. The results of these studies demonstrate the importance of information in pest management decision making.

Information concerning integrated pest management (IPM) techniques is also an important determinant of pesticide use levels. An examination of the use of IPM techniques undertaken by Wetzstein et al. (1985) determines that implementation of IPM does not have any impact on pesticide expenditure levels. That result is in direct contrast to studies by Antle (1988), Hall (1977) and Miranowski (1980) who all find evidence that IPM techniques help to reduce dependence on chemical pesticide use (Hall, 1977; Miranowski, 1980; Wetzstein et al., 1985; Antle, 1988). This paper incorporates both the information provided by pest control advisors and the information available on IPM techniques to determine the impacts of these factors on pesticide use decisions.

Information on pest management is available to growers from county extension agents, farm advisors, publicly available pest information forecasts and also from pest control advisors. Pest control advisors may be independent or they may be affiliated with pesticide companies and act as pesticide salesmen. Pest control advisors offer growers individualized services that usually can not be provided by extension specialists or public pest forecast and infestation information. Specifically, advisors are able to provide field specific information to growers, while public pest forecasts are generally given for entire regions and extension specialists are not always available and may not have the resources to provide field specific advice on a regular basis.

The analysis in this paper examines the role of pest control advisors in determining the use of insecticides to control fruit and army worm damage in late season processing tomatoes in California. This category of pesticides is of particular interest with respect to worker safety and there has been extensive

IPM research undertaken and programs have been established in an attempt to reduce chemical use. While early planting is recommended as a cultural management tool and could help to decrease pest populations, the planting date for tomatoes is typically set by tomato processors and thus is an exogenous factor to the decisions made by growers. Pest management programs for fruit and army worm control are commonly based on pesticides, although biological control alternatives are being tested by some growers (UC IPM) Thus, in this case, pest control is a decision as to whether to apply pesticides and if so, to what intensity. Both of these pesticide decisions can be based on IPM field monitoring and decision guidelines.

The IPM Group of the University of California (1998) has developed sampling procedures and thresholds for monitoring tomato fruit and army worm and they have also identified natural enemies of these pests in order to assist growers in reducing late applications of insecticides. Implementation of these IPM procedures has significantly decreased late-season insecticide use without a corresponding increase in worm damage (UC).

In this paper, IPM is defined as the quantitative and qualitative characteristics of monitoring for late insect pests and their natural enemies. Quantity of monitoring is measured as the time spent in the field inspecting and understanding the current pest situation and the quality of monitoring is represented as the ability of the grower to identify and consider natural enemies when making pesticide use decisions.

As mentioned above, pest control advisors can be affiliated with pesticide companies and there are several incentives for growers to use the advice of their pesticide salesmen rather than hiring an independent consultant. One reason is that pesticide salesmen maintain or have access to vast data bases which keep them informed about changes in pesticide markets and registration requirements. Therefore, salesmen are able to provide up to date information on pesticide markets, the appropriate choice of products for an observed pest problem, and the current status of new product registration. Even if a grower employs an independent consultant, he may still need to obtain this additional information from his pesticide salesman.

Another reason for employing a pesticide salesman has to do with current legislation in California which places liability on growers for all damages caused by pesticide use if that use violates the restrictions imposed by pesticide registration (CDFA). Many licensed pest control advisors will share this liability with growers when giving a written recommendation. Thus, the grower can partly insure his pest management for violation of regulation by using a licensed pest control advisor. Since pesticide salesmen typically hold licenses, but independent consultants and extension agents may not, this is an additional incentive for growers to use the recommendations of their pesticide salesman.

A third reason that pesticide salesmen are a popular choice has to do with the fee for monitoring and reporting that is included in the price of the chemical pesticide. When utilizing a salesman, the grower pays for the pesticide and the advising services jointly and therefore the services of salesmen are in essence free to growers since they do not have the alternative of paying less and receiving only the pesticide.

Given these advantages to using pesticide salesmen as advisors, there are not many growers in our sample who used independent pest control advisors and therefore the term pest control advisor is used here to refer to advisors who work for pesticide dealers. The next section introduces the conceptual model used to examine the incentives of growers and pest control advisors when making pesticide use decisions.

CONCEPTUAL MODEL

The monitoring activities of growers are characterized by the time allocated to monitoring and the quality of that monitoring. In this study, quality of monitoring is determined by growers using IPM guidelines to differentiate insect pests and natural enemies. The more involved growers are in monitoring, the more likely they are to be the first to identify pest problems and to identify them in the early stages when they can be controlled with fewer chemical inputs. Thus, individual growers may invest more in monitoring activities to both reduce damages and also reduce chemical pesticide costs.

It is assumed that if the monitoring activities conducted by either the pesticide salesman or the grower lead to identification of pest problems that require intervention, then the discoverer will suggest the amount of pesticide to be applied and the recommendation will be implemented. Let A denote the fraction of acres on which pest control advisors identify problems and recommend solutions, and $1 - A$ be the fraction on which the growers' own monitoring activities result in pesticide applications. The value of X_1 is the application per acre recommended by the grower and X_2 is the application per acre recommended by the pest control advisor, such that X, the observed application per acre, is

$$X = (1 - A)X_1 + AX_2. \quad (1)$$

Pesticide application levels recommended by growers are likely to be smaller when the grower is more involved in monitoring activities. In particular, it is reasonable to assume that X_1 is a function of growers' monitoring time, L, and monitoring quality, Q, as well as tomato price, P, such that, $X_1 = f(L, Q, P)$.

The change in the growers' recommended application level with respect to monitoring time, f_L, is assumed to be negative. This suggests that the more time devoted by growers to monitoring, the earlier they may detect pest problems and the fewer pesticides that will be required for treatment. Similarly, f_Q is assumed to be negative as increasing monitoring quality will also reduce the growers' own pesticide use recommendations. The change with respect to tomato price, f_P, is assumed to be positive because higher tomato prices are likely to make growers more cautious thereby encouraging them to spray more pesticides in order to reduce the risk of infestation.

A pest control advisor may not be aware of a grower's monitoring efforts but may be aware of the grower's education level and also the price of tomatoes. We assume advisor recommendations are a function of both of these factors, $X_2 = g\ (E, P)$. It is assumed that advisor recommendation levels increase with tomato price and decrease with the level of grower education. The level of grower education may affect the pest control advisors' recommendation for two reasons. First, the advisor may expect that educated growers are more aware of their field situation and therefore, may prescribe lower amounts of pesticides expecting that educated growers are better able to identify complications as well as better able to apply the chemicals appropriately. Second, since the advisor makes money from pesticide sales, he may sometimes over-prescribe, but he is more likely to over-prescribe to less-educated farmers. Thus we assume, g_E to be negative and g_P to be positive.

The growers' fraction of pesticide choices, $1 - A$, is assumed to increase as monitoring time and monitoring quality increase. Thus, A_Q and A_L are negative. The difference between the growers' and the advisors' recommendations, $c = X_2 - X_1$, can be interpreted as a cost of using the monitoring services of the advisors. We can not say with certainty that the value of c will be positive or negative, but *a priori* there are characteristics of the advisor-grower relationship which suggest that X_2 is typically greater than X_1.

First of all, advisors may receive some payment as a percentage of their chemical sales and therefore it is in their interest to suggest greater pesticide use. They obviously do not want to over apply to the extent that they lose clients, but they do have some incentive to suggest more chemical use than would the grower himself. Secondly, advisors want to retain a strong grower client base for future seasons and want to satisfy grower expectations with respect to crop damage. Therefore, advisors may tend to prescribe pesticides at higher levels in order to ensure that crop damage is limited to an acceptable level. Lastly, there is evidence in our survey that farmers are more likely to take a risk and use less pesticides if they are making the pesticide use decisions themselves. That is, there is a feeling of control associated with overseeing pesticide use. Given these characteristics, we expect to find that c is positive and that growers respond to an increase in c with an increase in their own monitoring efforts, thus making A_c negative.

The amount of time growers' spend monitoring has a cost equal to the opportunity cost of the grower's time at the time of that monitoring, which is represented as pm in this model. Growers' monitoring quality is determined by their knowledge concerning decision guidelines and this is assumed to be fixed during the growing season as it requires previous education and training. The opportunity cost of time for the training to acquire Q might be considerably lower than the opportunity cost of L as the grower can choose to learn during a period with low opportunity costs. For L, only costs during the growing season are relevant costs and these costs may be among the highest of the year. Therefore, L_{p_m} is likely to be negative.

We assume that once the grower has developed the ability to distinguish between pests and natural enemies, he will do so every time he monitors and thus, quality is a function of human capital, H. On the other hand, monitoring time depends to a greater extent on the field characteristics which influence expected damage from pests, E(d). Greater expected damage is assumed to increase grower monitoring time such that $L_{E(d)}$ is likely to be positive.

The next section describes the econometric model that is created to test the hypothesized relationships discussed in this section.

THE ECONOMETRIC MODEL

A primary objective in this analysis is to estimate the difference in the amount of pesticides recommended by growers and the amount recommended by pest control advisors. Unfortunately, data are only available for the aggregate amount of pesticides applied and it is therefore necessary for us to estimate the individual shares of this total. The relationship in Eq. (1) can be used to obtain individual shares according to the relationship

$$X = (1 - \hat{A})X_1 + \hat{A}X_2 + \varepsilon, \qquad (2)$$

where $\hat{A}$ is the predicted value for the fraction of acres on which advisors make recommendations and ε is a normally distributed error term (Just, Zilberman & Hochman, 1983). Estimation of Eq. (2) using the available data on total pesticide use and predicted values for A, yields parameters estimates for the individual pesticide use recommendations of growers and advisors.

The predicted values for A are estimated using the conceptual relationships discussed above. When conducting all estimation in this analysis it is important to consider the timing of decision making and the cause and effect relationships existing between the endogenous variables A, L, Q and X. If a simultaneous relationship between two or more of these variables exists, simultaneous equation bias will be a problem in our analysis. Bias will also occur if there is an additional unmeasured variable that is correlated with at least two of the endogenous variables in the model. Burrows estimates a system of equations representing pesticide demand and IPM adoption and comes to the conclusion that it may be impossible to obtain unique solutions for the endogenous variables in a simultaneous equation system unless restrictions are placed on the model (Burrows, 1983).

In this case, the grower's pest management decisions are not all made simultaneously. Since Q depends on human capital and is assumed to be acquired over some time period before the growing season, changes in its value affect L, A, and X but not visa versa. The variables L and A influence X, however, within one season, X does not affect either L or A as these variables are decisions made prior to X. There is endogeneity between the variables A and L as the level of advisor participation is a function of growers' monitoring time and the amount of time growers spend monitoring is a function of advisor participation. In order to account for this endogeneity, these variables are estimated as suggested by Burrows where the predicted values of L are used as an instrument in the estimation of A. Estimation of Q, L, and A is therefore defined as,

$$Q = \gamma_1 + \beta_1 H + \beta_2 t_1 + \nu_1 \tag{3}$$

$$L = \gamma_2 + \beta_3 \hat{Q} + \beta_4 p_m + \beta_5 E(d) + \beta_6 t_2 + \nu_2 \tag{4}$$

$$A = \gamma_3 + \beta_7 \hat{Q} + \beta_8 \hat{L} + \beta_9 E(d) + \beta_{10} t_3 + \nu_3, \tag{5}$$

where, $\beta_i t_i$ are vectors of parameter estimates and other factors that affect the dependent variables (these additional factors are discussed below) and ν_i are error terms. Figure 1 provides an illustration of the dependence of the factors that influence pesticide recommendations and pesticide use. Given these influences, the order of estimation is Eqs (3), (4) and then (5) with the resulting $\hat{A}$ being used in the estimation of Eq. (2).

Once estimates of grower and advisor pesticide recommendation levels are obtained, the difference between the two recommendations represent the cost of using an advisor, c. If c is positive, for the reasons outlined above, then increasing the share of the advisors' recommendation increases overall pesticide use. To test if an increase in the value of c leads to a decrease in advisor participation, c is included in Eq. (5) and A is estimated again. If c has

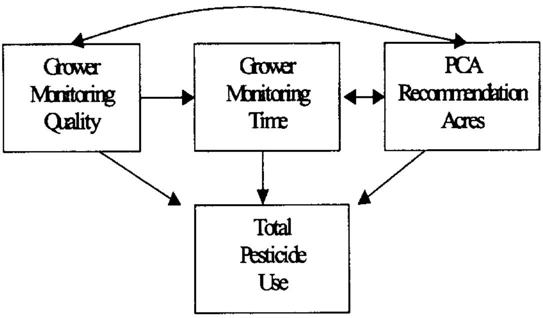

Fig. 1. Factors Which Influence Pesticide Use.

a negative significant effect on A, then this suggests that growers react to the advisors' higher pesticide use recommendation by reducing advisor participation in pest management decision making.

Finally, estimations are conducted to examine the effects of all factors on the levels of grower and advisor pesticide use recommendations.

$$X_1 = \gamma_4 + \beta_{11}\hat{Q} + \beta_{12} + \beta_{13}E(d) + \beta_{14}t_4 + \nu_4 \tag{6}$$

$$X_2 = \gamma_5 + \beta_{15}E(d) + \beta_{16}t_5 + \nu_5. \tag{7}$$

The next section provides detailed information on the data collected from the survey of California tomato growers and the other factors included in the estimations.

DATA

The methodology of the previous section was applied to data from a 1990 growing-season survey of 73 processing tomato growers in the six northern California counties of Colusa, Sutter, Sacramento, Yolo, Solano, and San Joaquin. Growers were interviewed to obtain information on the factors which influence their pesticide use decisions. Together, these growers worked 163 fields of processing tomatoes with total acreage of 59,658, or 19.2% of California's total tomato acreage and 16.8% of total U.S. acreage. The information collected in the survey concentrated on socioeconomic characteristics of the firm, the adoption of various cultural techniques of IPM, the quantity and quality of monitoring of different pest problems, and details concerning the output and inputs of tomato production.

Total pesticide use is examined as expenditure in dollars per acre. Other possible measures of this variable include pounds of active ingredient, number of acres treated, and number of applications per acre. Expenditures are chosen because pesticides are a heterogeneous group of inputs and the efficacy of pesticides seems to be better reflected by the expenditure in dollars per acre than by pesticide quantity which can vary substantially for different chemicals.

The advisors' participation rate is measured by growers' responses on the survey to estimates of their own and their advisors' share of monitoring. Survey results show that the share of participation of growers in monitoring is on average 65.8%. The value for growers' monitoring time is measured as growers' own estimates of their monitoring time in hours per acre, including employed personnel, spent monitoring for pest infestation. Data on monitoring quality is also collected from the survey and is a binary variable as to whether or not natural enemies are identified and considered in growers' decisions concerning pesticide use.

The explanatory variables that are common to the estimations of Q, L, A, X_1 and X_2 and are included in vectors t_1 through t_5 are education in terms of years of schooling as a measure of human capital, the price of tomatoes, a regional dummy for the two northern counties of Sutter and Colusa, and an estimate of expected damage. Expected damage is measured using the share of tomato production consisting of late season tomatoes, the harvest date of the field, and the observations on the infestation of early season insect pests. A higher share of late season production suggests a higher potential for damage while greater infestation in the early season suggests greater damage for late season production. The dummy variable for northern counties represents the fact that PCA fees in the northern counties are typically higher than in the southern counties.

Additional explanatory variables common to Q, L, and A and included in t_1 through t_3 are growers' ages in years, growers' estimates of opportunity costs of time during the growing season, total farm acreage, total farm income, and total tomato acreage. The share of total income resulting from tomato production is used as a risk proxy assuming that as this ratio increases, the grower will increase monitoring but is less likely to base pesticide application decisions solely on the observation of natural enemy populations.

A variable accounting for expenditures on seeds is included in t_2 and t_4 as a seasonal risk proxy for the more seasonal decisions of L and X_1. The grower has the choice of planting either hybrid or open pollinated varieties, the later being about four times more expensive on a per acre basis than the former. Seed expenditure is a major cost for producers and since pesticides are a risk-reducing input, they help to protect the grower's investment in more expensive seeds. The pest control advisor does not know a grower's expenditure on seed but only knows whether hybrid or open pollinated varieties are planted on the field. Therefore, a dummy variable for the use of hybrid varieties is included in t_5 when estimating X_2.

A weather variable, the sum of degrees above 90°F (i.e. degree days), is included in t_4 and t_5. The decision to spray pesticides is affected by high temperatures because both heat and pesticides can cause increased stress to tomato plants. For t_5, farm acreage and tomato acreage are included since the profit the advisor makes on one grower depends on the grower's acreage. For t_4, the risk proxy (the share of income from tomatoes as a share of total income) and the grower's opportunity costs of time are included.

A closer look at the nature of the dependent variables in the model is necessary to determine the appropriate estimation procedures to be used. In the case of pesticide recommendations, a certain number of growers may not apply pesticides on their fields at all. The observations of these zero recommendations have to be taken into account and therefore, a Tobit model is used here to calculate the elasticities of the predicted value of adoption as well as elasticities of changes in expenditures incurred after recommendation (Shakya and Flinn, 1985; Lutz et al., 1986; Cox & Ziemer, 1985). The variables for advisor participation, A, and grower monitoring time, L, are continuous variables with no truncation and so ordinary least square estimation is applied to Eqs (4) and (5). The binary variable for quality of monitoring, Q, is estimated using a logit model, which is the standard method in this case (Burrows, 1983; Malla, 1990; Lee & Trost, 1978).

ESTIMATION RESULTS

The results of the Tobit model estimating the pesticide use recommendations of growers and advisors are presented in Table 1. Most of the variables included in the estimation of the advisor pesticide use recommendations are significant at the 5% level. However, only a few of the variables are significant in the estimation of the grower recommendations. The advisor recommendation is significantly affected by expected damage measured by harvest date and early insect infestation, the education of the grower, tomato acreage, farm acreage, degree days and the grower's choice of hybrid or open pollinated varieties. The grower recommendation is significantly affected by the opportunity cost of the grower's time, seed expenditure, degree days and the regional dummy. With respect to the elasticity of the predicted probability of adoption and the change in expenditures, as presented in Table 2, a change in any of the explanatory variables has a greater effect on expenditures than on the probability of adoption.

Table 1. Growers' and PCAs' Recommendation of Late Applied Insecticide Use in Processing Tomatoes in Northern California.

Exogenous Variables	Grower's Late Insecticide Use Recommendation (X_1)	PCA's Late Insecticide Use Recommendation (X_2)
Intercept	0.3420	0.6580
Tomato Price	–0.0445	–0.0054
	(–0.57)	(–0.18)
Monitoring Time	–1.4418	–
	(–1.38)	
Monitoring Quality	–1.4086	–
	(–0.95)	
Opportunity Costs	–0.0509*	–
of Time	(–1.68)	
Education	0.2602	–0.3538***
	(1.06)	(–3.23)
Tomato Acreage	–	–0.0011***
		(–2.74)
Risk	0.0299	–
	(1.11)	
Early Infestation	–0.2657	0.8319***
	(–0.52)	(2.56)
Farm Acreage	–	0.0003**
		(2.37)
Regional Dummy	–2.612*	1.4778
	(–1.61)	(1.22)
Harvest Date	0.0287	0.1276**
	(0.22)	(2.32)
Seed Type	–	–0.9545**
		(–2.21)
Seed Expenditure	0.0137*	–
	(1.81)	
Degree Days	–0.0156**	0.0087**
	(–2.34)	(2.49)
Number	163	163
Average Pesticide Use Share in $/a	6.26	16.30
Average Pesticide Use Recommendation in $/a	20.96	26.25

t-values in parenthesis.
* = significant at 10%, ** = significant at 5%,*** = significant at 1%.

Table 2. Decomposition of Elasticity of the Expected Value of the Dependent Variable (Significant Variables).

	Grower's Elasticity of:		PCA's Elasticity of:	
Explanatory Variable	Pred. Prob. of Adoption	Change in Expenditure	Pred. Prob. of Adoption	Change in Expenditure
Harvest Date	–	–	3.12	3.73
Early Infestation	–	–	0.72	0.87
Degree Days	–0.47	–0.56	0.48	0.58
Regional Dummy	–0.32	–0.39	–	–
Education	–	–	3.61	–4.32
Opportunity Costs	–0.41	–0.49	–	–
Tomato Acreage	–	–	0.73	–0.87
Farm Acreage	–	–	0.60	0.72
Seed Type	–	–	–0.43	0.51
Seed Expenditures	0.39	0.47	–	–

Expected damage measured by the harvest date has a positive effect on the advisors' pesticide recommendations. A 10% change in the harvest date towards the end of the season increases the advisors' probability to use pesticides by 31.2% and the recommended pesticide expenditures increase by 37.3%. The level of growers' education has an even greater effect on advisor pesticide recommendations than does the harvest date. A 10% increase in a grower's education reduces the probability of the advisor recommendation by 36.1% and reduces expenditures by 43.5%. It is concluded that advisors' recommendations to spray and to what intensity are mostly determined by the calendar and by growers' knowledge.

Advisor recommendation is negatively related to tomato acreage and positively related to farm acreage. An increase of 10% in tomato acreage reduces advisor recommended expenditures by 8.7% while a 10% increase in farm acreage increases expenditures by 7.2%. We know that the revenues obtained by an advisor are dependent on the grower's total number of tomato acres. This result suggests that as tomato acreage increases, fewer chemicals must be sold to cover advisors' fixed costs, so advisor recommendations decrease. Farm acreage is interpreted as a measure of growers' capital and this has a positive effect on advisor recommendations as growers can afford greater pesticide use.

Seed expenditures are positively related to the grower's intended pesticide use. Pesticides are a risk-reducing input and therefore protect growers' investments in more expensive seeds. Correspondingly, our results conclude that a 10% increase in growers' seed expenditure increases pesticide use by 3.9% and expenditure by 4.7%.

Table 3 presents the results from the estimation of Eqs (3), (4), and (5). Advisor participation is significantly influenced by the two measures of expected damage, early insect infestation and the share of late-planted tomatoes, and it is shown that increases in the opportunity cost of growers' time leads to decreases in advisor participation rates. If both the growers' and the advisors' monitoring are equally productive, then one would assume a positive relationship between advisor participation and grower opportunity cost, especially since the pesticide salesman's time is without charge. However, if the grower assumes that he himself does a better job than the advisor, then grower opportunity cost is actually a measure of grower human capital and this relationship would be negative. Another possibility is the difficulty farmers may have providing a value for their opportunity cost of time and therefore this variable may be only an approximation of the true value. The change in advisor participation with respect to pesticide cost differences, A_c, is estimated as being significantly negative. Growers increase their own participation as the cost of following the advisors' recommendations increases.

The negative relationship between tomato acreage and the advisor participation rate is surprising as one would assume that an increase in tomato acreage would lead to an increase in advisor participation. There are two possible explanations. The first is that decisions in pest control are to some extent a matter of experience. A larger total acreage allows the observation of infestation events under different circumstances on different fields and such experience may lead to reduced reliance on advisors. A second explanation might be the fact that farms with larger tomato acreage employ their own production managers who can monitor the fields to a greater extent than can advisors who have additional tasks.

Predicted values of the growers' and advisors' use recommendations are computed and the average predicted share of the advisors' recommendation on pesticide use is calculated as $16.3 per acre while the average predicted share of the growers' recommendation on actual pesticide use is $6.2 per acre.[1] Eliminating the effect of the recommendation shares, pesticide use recommendations levels are calculated to be $26.4 per acre for advisors and $20.9 per acre for growers. Thus, advisors recommend, on average, pesticide use levels that are $5.5 per acre, or 26%, higher than growers.

It should be noted that the survey data used in this analysis are cross-sectional and therefore it may be the case that there are farm differences that are not captured by all of the variables from the survey. While the survey was

Table 3. Adoption of Qualitative and Quantitative Measures of Integrated Pest Management and the Participation of PCAs in Pest Management.

Exogenous Variables	IPM Monitoring Quality (Q)	IPM Monitoring Time (L)	PCA's share of recommendation (A)
Intercept	–0.5266	–0.8904	0.4613**
	(–0.18)	(–0.42)	(2.24)
Tomato Price	–0.07447**	–0.0124	0.0019
	(–2.47)	(–0.31)	(0.59)
Opportunity Costs	0.0384***	–0.0186	–0.0046***
of Time	(2.56)	(–0.92)	(–3.73)
Education	0.3432***	–0.0593	0.0029
	(3.02)	(–0.45)	(–44)
Age	0.0120	–0.0100	0.0023
	(0.68)	(–1.08)	(1.59)
Tomato Acreage	0.0003	–0.0003	–0.00009**
	(0.71)	(–0.90)	(–2.13)
Risk	–0.0217**	0.0283**	0.0036***
	(–2.03)	(2.30)	(2.85)
Farm Acreage	–0.0006***	0.0001	0.00003*
	(–2.59)	(0.62)	(1.82)
Farm Income	–0.4732**	0.0335	0.0017
	(–2.11)	(0.15)	(0.28)
Share of Late Tomatoes	0.3618	1.7396***	–0.2426**
	(0.37)	(3.25)	(–2.18)
Regional Dummy	–0.0008	–0.5293**	–0.1502***
	(–0.002)	(–2.48)	(–3.93)
Early Insect Infestation	–	0.0080	–0.0048**
		(–1.06)	(–2.01)
Seed Expenditure	–	0.0016	–
		(–0.91)	
Monitoring Quality	–	1.58	–0.0588
		(0.67)	(–0.70)
Monitoring Time	–	–	–0.0826
			(–1.60)
Pesticide Cost Difference (c)	–	–	–0.0008**
			(–1.98)
R^2 adjusted	0.20	0.15	0.35
	(Maddala R^2)		

t-values in parenthesis.

* = significant at 10%, ** = significant at 5%,*** = significant at 1%.

designed to capture the most important farm characteristics, there could be additional variables that help to explain some of the results presented above.

CONCLUSION

Tomato growers decide the amount of participation in pest management undertaken by pest control advisors, the level of adoption of IPM qualitative monitoring techniques, the amount of time spent monitoring, and the level of their own pesticide recommendations. This paper provides an econometric framework to estimate the factors that influence these decisions. Growers seek to minimize the costs of pest management (the costs of pesticides, monitoring, and yield losses) while pest control advisors employed by pesticide companies maximize profits through the sale of pesticides. Since the price of chemicals and the price of advising are not separable, the recommendations of pesticide salesmen are essentially free to growers, but on average these recommendations lead to higher pesticide use. The grower has to decide on the optimal participation of the advisor, thus making a trade-off between the costs of his own participation and the higher pesticide expenditure incurred if following an advisor's recommendations.

Two reasons are put forth to explain why advisors' recommendations are, on average, higher than those of growers. One is the higher intensity and, therefore, accuracy of the monitoring of growers which leads to lower pesticide use. Another suggested explanation is that advisors maximize their profits through product sales. It is suggested that a grower's willingness to accept these higher expenses results from the perception of pesticide salesmen's advice as free, the transaction costs entailed in choosing another advisor, and a lack of market transparency. If prices for pesticide products and the service of advisors were charged separately, growers might be less willing to accept recommendations for higher pesticide use. Also, market transparency would increase if growers could compare the prices for products separately from services. In fact, agricultural cooperatives in some areas charge a fixed per acre fee for services and sell pesticide products separately.

In this study of California tomato production, advisor recommended pesticide use levels are largely determined by the expected harvest date of tomatoes. Harvest dates are set by the processor and are therefore a function of processing capacity. Note that this analysis suggests that a significant factor influencing pesticide use is actually outside the control of the grower. This is an important observation because any restricted use regulation which is imposed to limit growers' use of pesticides does not directly address this particular incentive, which is actually created by processing constraints.

Results also suggest that growers depend heavily on the information obtained from directly observing their fields. The more time spent monitoring and the higher the quality of that monitoring, the less chemical pesticides the grower feels are needed to avoid crop damage. This suggests that IPM programs, which encourage monitoring and provide knowledge on what growers should look for when monitoring, can be successful in reducing the amount of chemical pesticides used in pest management.

The perception of growers' education also greatly affects advisors' pesticide use recommendations. More pesticides are suggested for use on fields where growers are perceived to be less informed as to pest management decisions. Therefore, increasing the perception of grower education through improved training would help to reduce chemical pesticide use. Furthermore, such training is likely to increase the probability of farmers adopting IPM qualitative monitoring guidelines because the level of human capital is an important factor in the adoption process. Improved training could also change the current perception of many growers that IPM qualitative guidelines are risk increasing. This would lead to increased IPM adoption and lower chemical pesticide use.

NOTE

1. $$E(x_i) = \sigma[\beta' x_i \Phi(\beta' x_i) + \phi(\beta' x_i)]$$

where E( ) is expected value and Φ and ϕ are the cumulative and probability distribution functions respectively for the normal distribution (White, 1978).

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6. ORGANIC FARMING DEVELOPMENT IN EUROPE – IMPACTS OF REGULATION AND INSTITUTIONAL DIVERSITY

Johannes Michelsen

ABSTRACT

In Europe the distribution of organic farming has increased along with growing political support during the 1990s – including a common EU definition of organic farming and financial support for organic farmers. Three qualitative analyses covering all EU member and three non-member states are summarised to analyse co-variation between policies and organic sector size. When comparing impacts of policy instruments, the results were unclear but mainly pointed towards positive effects from introducing uniform certification schemes. With regard to national policy processes no correlation appeared between conditions for policy oriented learning and the size of organic farming sector. Some explanatory power is, however, derived from distinguishing between three types of institutional interrelationships between organic farming and mainstream farming. Cooperation or creative conflict persists in countries with large organic farming sectors, while pure competition is found in countries with small ones.

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INTRODUCTION

Organic farming developed quite fast during the 1990s from covering shares of total agriculture just above zero all over Europe to reaching quite substantial levels in Nordic and German speaking countries where organic farming in a few years has reached shares up to above 10% of the total utilized area, the EU average being about 2% (Lampkin & Foster, 2000). Even in Italy organic farming has reached a substantial share of total agriculture. During the same period consumer demand for organic food has increased throughout Northern Europe (Michelsen et al., 1999). Because of organic farming's basis in a fundamental criticism of modern mainstream agriculture, the growing public interest in organic farming and food may be seen as representing within the European food sector the general search by the end the 20th century for alternatives to those production methods associated with problems with regard to environment, public health, animal welfare and similar issues (Michelsen, 2001a).

Organic farming is, however, not a new invention of the food industry. It dates back to the work of individuals such as Rudolph Steiner in Austria in the 1920s and Lady Eve Balfour in Britain in the 1940s. Their work gained worldwide attention during the 1980s and 1990s in social movements as possible responses to the search for alternative farming methods, which combined concern for the environment and animal welfare with concern for the economic and social well being of farmers and consumers (Michelsen et al., 2001).

The interest in alternative farming methods manifested itself from the first half of the 1980s in political realms in European countries through political recognitions of organic production methods (certification) and introduction of (limited schemes of) public financial support for organic farmers. Eventually, common EU regulations on certification and financial support were introduced in 1991–1992 (Lampkin et al., 1999). The growing political attention to the issue is further emphasised by the political processes taking place in the first half of 2001. In January, Germany made a quick promotion of organic farming part of the political initiatives countering BSE and 19th of June the European (i.e. EU) Council of agriculture ministers decided to head for a European action plan promoting organic farming prepared by a conference held in May.

The development of organic farming is open to several types of analysis. The approach followed here is a policy analysis of possible policy impacts on growth in the attempt to give a first overview of the general development in Europe. This is done through comparing organic farming development within the general institutional environment of agriculture and food production in up to 18 European countries. The main reason for this choice is that it appears essential that organic farming has been able to introduce environmental and other politically salient concerns into an industry – agriculture – notorious for its resistance to influence from outside interests not least with regard to environmental regulation and demands for budget cuts. Some emphasis is put here on aspects of environmental regulation, as EU support for organic farming is part of the agri-environmental measures (EEC Regulation 2078/92).

In attempting to cope with the growth of organic farming, two competing lines of argument may be maintained, parallel to the discussion whether the aim of environmental regulation should remain a separate type of environmental policy or should become integrated into sector policies. One line of argument is that the types of policy instruments used in support of organic farming has mattered in the sense that organic farming growth seems caused by the introduction of policy instruments in support of organic farming. The other line of argument is that organic farming growth reflects a growing interest among farmers to integrate environmental concern into a distinct production system rather than accept environmental regulation that just add demands that may be more or less incompatible with existing farming methods. The discussion also implies considerations of organic farming being an example of self-regulation.

Available data on organic farming development are scarce and insufficient for attempts to reformulate the lines of arguments to competing hypotheses and test their explanatory power empirically. Hence, after a presentation of organic farming within a European context a theoretical framework is developed that characterises organic farming and its institutional interrelationships with general agriculture institutions and politics in order to set the stage for the actors involved in organic farming development. To this is added a discussion of different types of policy instruments available for actors involved in promoting organic farming. The empirical analysis is via comparison of the development in a varying number of European countries focused on identifying possible impacts on growth originating in policy instruments or different types of institutional interrelationships. Finally, the conclusion includes a summary of the findings in terms of their support to the two lines of argument.

ORGANIC FARMING PRINCIPLES AND THE EUROPEAN CONTEXT

As it may, from the outset, seem exhaustive to try to analyse the growth of one distinct type of agriculture in up to 18 countries which are so different with respect to the natural and social conditions for agriculture as the 18 central and

Western European countries analysed here, the purpose of this section is to demonstrate the relevance of the aim. After an introduction to organic farming from a social science point of view follows a short description of organic farming's position within European agriculture based on available statistics along with an introduction to the political environment in which organic farming policy has developed.

Principles: Criticism and Social Movement

The very definition of organic farming involves a fundamental criticism of mainstream agriculture. This is implied when Lampkin (1994) defines organic farming

> (. . .) as an approach to agriculture where the aim is:
>
> to create integrated, humane, environmentally and economically sustainable agricultural production systems, which maximise reliance on farm-derived renewable sources and the management of ecological and biological processes and interactions, so as to provide acceptable levels of crop, livestock and human nutrition, protection from pests and diseases, and an appropriate return to the human and other resources employed (Lampkin, 1994, 4ff).

The main reason why this definition is meaningful is that the reader knows beforehand that it stands as a nearly complete opposite to the viewpoints of mainstream agriculture which imply extensive use of artificial inputs such as fertilizers, pesticides etc. designed to increase productivity in food production.

Organic farming is based on distinct values as mentioned in the words of the Principle Aims of the International Federation of Organic Agriculture Movements (IFOAM) that involves a clear vision of a major change in society in order to make it possible

> (. . .) to interact in a constructive and life-enhancing way with natural systems and cycles; (. . .) to consider the wider social and ecological impact of the organic production and processing system; (. . .) to progress toward an entire production, processing and distribution chain, which is both socially just and ecologically responsible (IFOAM, 2000).

The value basis of organic farming and the strong criticism of mainstream agriculture implies a first point in considering organic farming as a social movement – and it is reinforced by a second point *viz.* organic farming has been developed by cooperative efforts of members of several social groupings, which either stand in the periphery of mainstream agriculture or are fully excluded from those dominating the agriculture sector (see Tovey, 1997; Padel, 2001; Reed, 2001). At least at the outset, it seems hard to maintain that these different groupings developed organic farming on the basis of common material interests. It seems more meaningful to see the cooperation of the different groupings and the formation of a social movement as based on *certain shared* (soft) *values* which in central aspects differ strongly from those values expressed by actors and organisations of mainstream agriculture (Michelsen, 1997, 2001a).

What distinguishes organic farming organisations from social movements (Tarrow, 1994) is, however, that organic farming – in addition to criticism – presents a positive definition of a better farming system and is able to demonstrate its value both in the field practises and in the food market (Lampkin et al., 1999). The market-oriented aspects of organic farming prompted at an early stage of organic farming development theories stating that organic farming is bound to loose its social movement identity because of an agri-business take-over (Buck et al., 1997). However, Michelsen (2001a), based on European experience, and Campbell and Liepins (2001), based on world market experience from New Zealand, demonstrate that these analyses are far from being confirmed in more developed organic farming sectors. It seems for instance that the way the organic farming production standards has worked up to now has implied strong needs for interrelationships where food companies rely heavily on farmers' ability to define and implement standards by themselves.

Organic Farming Within European Agriculture

During the last 20 years, organic farming in Europe has gone through a major change. In the 1980s organic farming was an obscure type of farming, practiced by very few farmers and relating with small and marginal groups in society with a special interest in this type of production. In 2001, however, organic farming in Europe is characterised by strong growth and in some countries representing a major dynamics in both agriculture production and the food market (Michelsen, 2001a; Michelsen et al., 1999).

Because of the lack of official statistics concerning organic farming, the safest and best reconfirmed account available of the number and size of organic farms in Europe is included in Foster and Lampkin (2000). It documents the development from 1993 to 1998 for all 15 EU member states along with Norway, Switzerland and the Czech Republic (non-member states of the EU). These statistics show that in 1993 about 0.5% of all holdings in the 18 countries were organic and they covered 0.7% of the total, utilized area (UAA). By 1998 these shares had tripled to 122,892 holdings (1.7% of all) covering nearly 3 mil. ha (2.2% of total UAA) and total growth has continued ever since. Nordic countries such as Sweden (15.0% holdings and 7.8% UAA) and Denmark

(3.5% holdings and 3.7% UAA) and German speaking countries such as Austria (9.6% holdings and 8.4% UAA) and Switzerland (6.2% holdings and 7.3% UAA) have large organic farming sectors whereas Mediterranean countries (other than Italy with 1.8% holdings and 5.3% UAA) and Franco- and Anglophone countries have small sectors covering less than 1% of both holdings and UAA. It thus appears that organic farming is not reserved to distinct climatic conditions or types of soil, but appears more attractive in some socio-cultural environments than in others.

In total, the average size of organic farms (24.3 ha) is a bit larger than the average of all European farms (18.8 ha). When evaluating the farm size it should be born in mind that it is a special objective for many parts of agriculture policy in Europe to keep individual farmers as the owners of farms and to postpone structural changes in the farm size. Regarding the size of European organic farms relative to all farms it is indicated by the figures mentioned, that there are major variations. In some countries the average organic farm is larger than the national average of farms while in others countries the average organic farm is smaller. This hints to the fact that organic farming is developing quite differently in different countries. In some countries organic milk production is very important while in other countries organic cereals, vegetables or other plant products are the most important ones.

The European Political Context

Lampkin et al. (1999) document the regulatory aspects of the organic farming development in Europe. It started in the first half of the 1980s with national recognitions of the production standards and certification systems and it was in some cases accompanied by financial support for developing certification, marketing, research or other aspects. National financial support for organic farmers was introduced in the latter half of the 1980s in a few countries. At the same time the political interest in organic farming support had moved to the level of the EU, who introduced a common set of production standards for organic plant production in 1991 (EEC Regulation 2092/91) (extended 1999 with common standards for organic livestock production (EEC Regulation 1804/99)) – and an option for financial support of organic farmers as part of the measures accompanying the reform of the Common Agriculture Policy in 1992 (EEC Regulation 2078/92). This position within EU regulations was confirmed in 1999 when organic farming support was maintained under the Agenda 2000 measures of agriculture policy. In European countries outside the EU organic farming has obtained a similar status.

Within the EU (and largely also in the other European countries) organic farming support developed under two main headings: consumer protection and agriculture policy. Regulations with regard to certification specify some common minimum demands for products sold as organic and inspection of compliance while leaving implementation to national political systems. The justification for regulating organic certification is consumer protection and the form is purely legal regulation. The other part of the policy in support of organic farming involves financial support to farmers and to organisations and agencies that promote organic farming and is related to agriculture policy. The first country to introduce financial support to organic farmers was Denmark where it was included into general market-oriented attempts to try to satisfy clear consumer demands for organic food (Michelsen, 2001b; Lynggaard, 2001a), while other countries, such as Austria and Germany took an agri-environmental approach.

In the EU context, organic farming financial support became part of a general agri-environmental support scheme, which was the major result of the attempts to reform the Common Agriculture Policy (CAP) (i.e. reduce the budget and change from paying subsidies to prices to paying subsidies on a per hectare basis) in 1992 (Whitby (Ed.), 1996). Support for organic farming was only one among more agri-environmental initiatives included in EEC Regulation 2078/92. This regulation demanded that some support for organic farming should be found in each member state and promised EU support to cover parts of the expenses, but decisions with regard to the level of support paid and the conditions to be met by applicants were left to national decision-makers and thus differs strongly between countries. There is, however, no clear correlation between the level of financial support and the national share of organic farming (Michelsen et al., 2001).

Whereas certification could be seen as a relatively uncontroversial issue in terms of agriculture policy, agri-environmental issues were – and still are – in fact controversial because they imply that agriculture has to meet specific environmental demands. This runs counter to the attitudes held by the very strong and influential organisations of agriculture both on the national and the EU level. They have developed tight and well-established interrelationships both among themselves, the food industries and the public sector during the last century. Agriculture is one of the sectors of society most heavily segmented from the rest of society. In the current situation agriculture in Europe has both become a strongly subsidised sector and a sector in which political decision-making to a major extent is characterised by self-rule – although with some differences between countries (Daugbjerg, 1998; Lowe et al., 2000; Winter, 1996). The controversies over agri-environmental regulation imply that agriculture rather than being regulated by the polluter-pays-principle has got options to obtain financial support for developing environmentally friendly production methods. The persistence of controversies over agri-environmental regulation is reflected in the fact that only small scale and marginal growth in environmentally friendly practices have occurred up to now. The growth of organic farming is the most prominent exception to this picture (Lampkin et al., 1999; Michelsen, 2001a).

The above overview of the European context of organic farming development may be summarised by stating that organic farming is distinguished from mainstream farming on a political and ideological basis. Organic farms are, however, not clearly distinguished from other types of farms in terms of size or production, and the regional distribution seems not clearly affected by specific technical or climatic issues. The political context in Europe is based on traditions of agriculture self rule where organic farming growth represents one of the few successes of relatively limited attempts to introduce environmental regulation into the agriculture policy. It hereby appears relevant to analyse organic farming as a political and social phenomenon rather than as mainly an issue of agriculture. Following this perspective, the focus must be on impact of policy on the aggregate number for organic farmers. Therefore, the empirical analyses that follow will focus on explaining the growth in the number of farms – as an approximation to the number of farmers.

ANALYTICAL PERSPECTIVES ON ORGANIC FARMING GROWTH

Social science analyses of organic farming development are rare.[1] Michelsen (2001a, b; Michelsen et al., 2001) suggests studies of organic farming development be based on an institutional approach that defines institutions, such as organic farming or mainstream agriculture, by their values, norms and rules, which might – or might not – be formalised into different types of organisation following March and Olsen (1989), Sjöstrand (1993) and Peters (1999) (which include a recent overview of institutional theories). The reason is that organic farming is distinguished from mainstream agriculture – or what organic farming has successfully been able to name 'conventional' farming – on exactly these aspects. Institutional theory, however, has major difficulties, in coping with change, such as organic farming growth, because values, norms and rules are expected to continue to reign in spite of all deliberate attempts to change them (Brunsson & Olsen, 1993). Hence, within an institutional theoretical approach it becomes a specially analytical issue to cope with changes in the institution of mainstream agriculture and the dynamics of introducing organic farming into the agriculture sector.

Networks and Advocacy Coalitions

Existing analyses of agriculture policy reform and agri-environmental politics apply network theory (Daugbjerg, 1998) or other theories with some emphasis on organisational or institutional aspects although not necessarily on the basis of institutional theory mentioned above. However, these analyses appear to have some difficulties in explaining dynamic processes such as the reforms of the CAP as they build on a rather static view on the field of agriculture policy and often explain major changes as the result of external influence, such as changes in socio-economic conditions or world politics such as the WTO (Lynggaard, 2001b).

Considering the development of organic farming, the theoretical contributions of Sabatier (1993) and Jenkins-Smith and Sabatier (1993) appear relevant. They combine network theory with an analytical interest in the basis of a policy in values/beliefs. Sabatier's (1993) 'advocacy coalition framework' implies that individuals originating in different organisations or parts of society may share distinct sets of beliefs ranging from core beliefs resistant to change to secondary beliefs which might be changed during processes of 'policy oriented learning'. These shared beliefs form the basis for different advocacy coalitions that influence politics within a demarcated political subsystem on the basis of exchange of arguments regarding beliefs. Such a framework accords well with the above perception of organic farming as a social movement criticising mainstream agriculture on the basis of values.

Sabatier (1993, p. 13) emphasises that policy change should be analysed "over a decade or more" and Jenkins-Smith and Sabatier (1993) specify conditions under which long-term policy change may take place in terms of learning. The theory thus points to a need for searching for advocacy coalitions representing beliefs of organic farming and mainstream farming respectively and to detecting their relative influence on the long term development of organic farming policy.

Sabatier admits that attempts to change policy may be off set by interrelations between advocacy coalitions characterised by dominance rather than exchange of arguments and beliefs and that "external perturbation" (Sabatier 1993, p. 34) rather than processes internal to a policy subsystem may cause major changes of policy. In terms of the development of organic farming this translates into discussing the possible dominance of mainstream agriculture advocacy coalitions and networks over those promoting organic farming and the possible introduction of external forces in support of organic farming. None of these approaches seem, however, of interest for the analysis made here.

With regard to power mainstream agriculture organisations are so dominant within the agriculture policy subsystem that the weak and deviant organic farming coalitions on all measures of power are far from being able to match them. Neither the promotion of organic farming has been backed anywhere in Europe with the power of anything similar to an external perturbation.

When neither the options of power relations or external perturbation prevail, Jenkins-Smith and Sabatier (1993, pp. 48–55) specify that the probability for policy change via processes of value based policy-oriented learning may be influenced by three variables. These variables include:

The level of conflict – productive dialogue or learning is less likely the less compatible are the core values of competing coalitions. Hence, learning presupposes "an intermediate level of informed conflict" (Jenkins-Smith & Sabatier, 1993, p. 50).

Analytical tractability – learning is less likely the larger the disagreement on how to analyse single issues.

The nature of the forum for discussion between advocacy coalitions – learning is more likely the more prestigious or attractive the forum is to representatives of different coalitions and the more it is dominated by professional norms.

These variables form the basis for the attempt here, to make an institutional explanation of the relative success of organic farming development within the strongly organised agriculture segment in some European countries. However, some adoption is needed for the theory to fit the special circumstances of organic farming development. Especially with regard to the first variable, level of conflict, it seems needed to introduce a qualitatively other way of dealing with conflict than the one offered by Sabatier and Jenkins-Smith.

Cooperation, Competition and Creative Conflict

Regarding the level of conflict, a scale ranging from low via medium to high level of conflict may be figured out. However, such a scale may disguise important qualitative differences not least when considering organic farming, which attempts to introduce new issues and a new way of dealing with them to the agenda of the agriculture policy subsystem. In the case of a low level of conflict, the values expressed by organic farming are clearly not perceived as a threat to mainstream agriculture and hence conflict may nearly seem non-existing. In this instance it even seems more relevant to talk about cooperation than conflict between organic and mainstream agriculture. In the opposite case of high level of conflict, the values of organic farming are strongly opposed by mainstream agriculture and conflict may reach a level of hostility involving attempts from both parties to damage the position of the adversary in the eyes of the public. This is in other words a situation, which seems very similar to strong market competition. In this way, the extremes of the scale of conflict may be defined as pure cooperation and pure competition, which in turn correspond with the contrasting of competition (market) and cooperation (association) as organising principles in institutional theory of organisations (see for instance Sjöstrand, 1985, 1993; Michelsen, 1994).

When considering the level of conflict, it is in line with Jenkins-Smith and Sabatier to make special attention to the middle point in-between the extremes emphasising that conflict should not be too strong when attempting to resolve it. Within the cooperation-competition framework introduced here, however, the interest in the middle point also involves a major theoretical interest, as it must involve some combination or a mixture of cooperation and conflict. Such a position is not discussed very often within institutional theory (see Michelsen, 1994). However, it seems evident that in terms of cooperation-competition an intermediate level of conflict between organic farming and mainstream agriculture is not easy to obtain and that such a situation may seem rather unstable. The suggestion for the analysis done here is to characterise the intermediate level of conflict between mainstream and organic agriculture as *creative conflict* implying cooperation in some areas and competition in others – and to add the option that interrelations for any given area may be subject to change over time. Thus, regarding the level of conflict, three types of interrelationship with strongly diverging qualities may be distinguished. Their properties are discussed below and summarised in Table 1 (Michelsen et al., 2001).

Pure co-operation is one extreme type of interrelationship between organic farming and general agriculture institutions. Pure co-operation is a situation where cooperation between the two parties is so comprehensive and encompassing that the fundamental conflict inherent in organic farming's criticism of mainstream farming is avoided and deliberately toned down to such an extent that the difference between the two nearly seems to disappear. There may be several reasons for avoiding or toning down conflicts. One is the conviction that organic farming more or less equals existing types of (extensive) farming, another that organic farmers are not that different from other farmers. In such a case it is very difficult to maintain the distinctiveness of organic farming – the identity may wither away – and one should expect

Table 1. Three Main Types of Institutional Interrelationship between Organic Farming and General Agriculture Institutions.

Characteristics	Pure co-operation	Creative conflict	Pure competition
Contact between organic and general agriculture institutions	Comprehensive and encompassing cooperation in all aspects	Co-operation in some aspects and competition in other aspects	No contact at all
Need of organic farming organisations	No	Yes	Yes
Perception of interests	Silence on differences in farming systems	Joint perception of some interests – for instance regarding the environment – opposing perception of other aspects – for instance GMO	Suppression of all interests and arguments of the adversary
Exchange of views	Differences toned down	Competition and mutual respect for others' views	No serious attempts for exchange
Expected consequences for organic farming identity	Wither away	Established but development on pragmatic basis	No change
Expected consequences for dissemination of organic farming	No continuous and substantial development of organic farming – unless if perceived as future for all national agriculture	Organic farming develops stepwise based on creative solutions to issues of co-operation or competition	Organic farming development hampered

Source: Michelsen et al. 2001.

only to find few and comparatively weak organisations that exclusively forward the ideas and interests of organic farming. Instead, it is expected to find the main proponents of organic farming inside mainstream agriculture institutions. Pure cooperation is not expected to promote a continuing and substantial dissemination of organic farming – unless in a situation where organic farming is perceived as the future option for all national agriculture.

The other extreme type of interrelationship is *pure competition.* It is characterised by none or only occasional direct contact between organic farming institutions and those of general agriculture because they see each other as competitors or opponents *vis-à-vis* the food market, public agriculture support or the public opinion rather than as farmer colleagues. In all domains, pure competition may create an atmosphere where attempts are made to suppress the interests and arguments of the adversary without any serious effort to exchange views on – for instance – perceptions on agriculture or farmers' strategies for action in the individual farm and in relation to politics and the market. Hence, a sense of 'fundamentalism' regarding both farming systems may develop – or what Jenkins-Smith and Sabatier term "dialogue of the deaf". Pure competition presupposes the existence of autonomous organic farming organisations. Competition will be open if the organic farming organisations are strong enough to be considered a real organisational obstacle to the general agriculture organisations. If the organic farming organisations are weaker, it may lead the general agriculture organisations to neglect them. The pure competition interrelationship is expected to hamper the development of the weaker part and hence to have a negative impact on organic farming growth.

Creative conflict is the type of interrelationship that lies in-between competition and co-operation. Here, organic and general agriculture institutions are in continuous contact while cooperating on some issues and competing on others. Hence, creative conflict may involve a climate of both competition and mutual respect under a joint perception of some – but not all – common interests – for instance regarding the development of agriculture in an environmentally friendly and economically sound way. This type of interrelationship presupposes the existence of distinct organic farming organisations. Creative conflict is, furthermore, expected to help in promoting the development of organic farming by keeping issues of organic farming on the agenda of the farming community, the food market and agriculture policy as well as in society at large, whilst maintaining the integrity of the core principles within a pragmatic framework. The conflict should be perceived as creative, not only for organic farming, but also for mainstream agriculture for instance in easing the ability of general agriculture institutions to develop environmentally friendly agriculture and to service new groups of farmers.

The three types of interrelationships are based on theory and represent three positions on a scale. Therefore real-world interrelationships may combine elements of two of the positions and hence be positioned somewhere between the three main positions.

It appears from the listing of the three positions that only creative-conflict-interrelationships, in which competition and co-operation are combined, is expected to contribute to the promotion of organic farming. In spite of their fundamental differences, both pure-competition- and pure-co-operation-interrelationships are expected to hamper organic farming growth and at the same time organic farming is thought to be in danger of loosing its identity. In terms of change both pure cooperation and pure competition may be perceived as stable types of interrelationship unable to bring about fundamental change. Hence, within this conceptual framework organic farming development must involve a shift from one of the two stable states to a period characterised by instability or dynamics, represented here by creative conflict. In this way, the theoretical discussion of the level of conflict has turned the analytical attention towards the dynamics of institutional change.

Regarding the second policy-learning variable, analytical tractability, the position of organic farming seems from the outset quite intractable for mainstream agriculture. Organic farming's pure denial of artificial inputs for reasons not based on scientific evidence regarding residues in food or similar arguments and the explicit focus on other aspects than productivity seems quite incompatible with the belief system of general agriculture institutions (Wynen, 1996). On the other hand organic farming seems in accord with the discussion of sustainability (Soerensen & Kristensen, 1992; Pugliese, 2001), which has gained importance within European agriculture policy. Hence, it may not be impossible to develop a conceptual platform common to organic and mainstream agriculture in order to improve the analytical tractability. A common platform seems, however, to presuppose quite substantial and deliberate action from several actors. To obtain that would be to fulfil an important part of managing the fundamental conflict between organic and mainstream agriculture. Hence, the issue of analytical tractability need not be treated as a separate issue but is included as an aspect of the three types of interrelationships.

The third variable of policy learning is about having a forum for discussion that is sufficiently prestigious and attractive to main proponents of all important advocacy coalitions. Such a forum may also be an aspect of conflict management as it involves the establishment of a platform for dialogue between parties, which from the outset must be considered adversaries. Such a forum (or several fora) may help developing paths of analytical tractability or joint solutions to political, economic or social problems involved when placing organic farming among other – more mainstream – types of agriculture. It may also help to connect actors and organisations of organic farming with those within mainstream agriculture in order to solve practical problems regarding for instance advice (farming community), certification procedures (agriculture policy) or sales channels (food market).

From this discussion it follows that the character of the conflict is expected to have major impacts on the development of organic farming. Development may both be hampered by too much cooperation and too strong competition whereas the most fertile climate for organic farming development seems to be characterised by creative conflict including cooperation in some fields and competition in others. In this way there seems to be a need for managing the level of conflict if organic farming is to develop. This task may be eased by means of establishing one or more fora for settling discussions between main actors of both organic and mainstream farming.

Organic Farming and Policy Instruments

Organic farming is a farming system developed on the basis of some fundamental values as expressed in the Principle Aims of the IFOAM (IFOAM, 2000). The declaration includes 17 messages that point in different directions and are not always easy to combine, but they form the basis for production standards, which are much more specific and action oriented. Production standards include statements about the non-allowance of using artificial fertiliser and pesticides, conditions for conversion etc. The values of organic farming thus have a more enduring status than the production standards, which in principle are only temporary attempts to realise values. The production standards gives organic farming a clear definition and serve as the basis for certification of organic farming and food production. Considered in this way, organic farming constitutes an example of pure self-regulation with production standards serving as the main regulatory instrument. Within such a system, the incentive to certify production remains with market conditions – the option of earning a price premium for certified organic products to cover possible extra production and other costs and some extra profits.

When considering options available for political support of promoting organic farming, it is common to distinguish between three main types of instruments (see Peters & Nispen (Eds), 1998 for a critical discussion). There is broad agreement as to the content of two of them: *legal instruments* (regulation) and *financial instruments* (economic (dis)incentives) both of which are functioning with government as the main actor while citizens are mere objects for intervention. Legal regulation operates through political power/authority and legitimate legal sanctions related to the state monopoly of power. Financial instruments operate through economic incentives whether positive in the form of support or negative in the form of taxes and duties, i.e. media related to the working of the market economy.

The third category of policy instruments includes soft and less clear instruments but has no clear labelling. Vedung (1997) suggests 'information' indicating a one-way flow of messages from public agencies to citizens while Dabbert (1997) and Parsons (1995) suggest 'moral suasion' that opens some space for citizens to consider their personal preferences related to the messages. de Bruijn and Hufen (1998) label the third category 'communicative instruments', leaving some space for a two-way interrelationship between regulator and regulated citizens. The third category thus involves some kind of interaction between the state and the private citizens and it is clear that – irrespective of the label – the effect of the instruments of the third category does not rest on clear (positive or negative) sanctions but on compatibility with views and attitudes held by the regulated citizens. Against this background Michelsen (2001b) argues that the third category is fundamentally about self-regulation.

In this way each of the three main types of policy instruments is related to one of the three main sections of society: legal instruments/regulation are based on the authority and power of the state; financial instruments/economic incentives are based on the price mechanism relating to the market; and communicative instruments/self-regulation are based on the mutuality and social norms of the civil society. Thus, a distinct mechanism of self-adjustment is at work within each section of society, but separate policy instruments adaptable to all sections are available if public intervention is demanded for some reason. In relation to organic farming development all three main sections are open to influence and political support may rely on intervention by instruments belonging to one or all of them.

The analysis of policy instruments suggested here may at first glance seem disturbed by Hofer's (2000) analysis of the development of organic farming in three EU countries, as she perceives the general development of organic farming a case of Joint Environmental Policy Making (JEP). JEP is defined as a type of voluntary regulation based on negotiations between actors of the public and private sectors and is distinguished from obligatory regulation and self-regulation respectively (Mol, Liefferink & Lauber, 2000, 3ff). Hofer rightly points out that the EU policy in support or organic farming is voluntary, but the distinction between voluntary and obligatory regulation should not be seen as constitutive for separate types of policy instrument. It rather represents a scale of coercion available when considering the use of any of the three types of policy instruments mentioned above. The conclusion of Hofer's empirical analysis of the development in Austria, Denmark and the Netherlands is that the relative success of organic farming in the two former countries is connected to a successful development of separate organic farming policy networks. This finding does not confirm that JEP is a separate policy instrument, but it points to the possible importance of policy networks for the proper functioning of a policy instrument and this confirms the relevance of the theoretical considerations above regarding policy-oriented learning.

Summarising the discussion so far on policy instruments in relation to organic farming development, the EU regulation on organic farming is characterised by including all three types of policy instruments. The EU certification system standard involves legal regulation while financial instruments are used to support organic farmers and associated industries – and financial support may eventually also be used in support of communicative types of policy instruments such as research, information and advice. Furthermore, all regulation is voluntary to citizens provided they comply with specific conditions and in most EU member states the policy instruments exist parallel with organic farming's own systems of self-regulation on which the proper functioning of the public policy may even depend.

EMPIRICAL STUDIES OF ORGANIC FARMING DEVELOPMENT IN EUROPE

The empirical studies presented are to contribute to explaining the development of organic farming under very different conditions throughout Europe. They cover three different but interconnected discussions and are based on three different sources originating in the same EU sponsored project.[2] The order of discussions is to move from a rather crude overview of the direct impacts of policy instruments on organic farming growth in 18 European countries to an introductory, institutional discussion about the presence of advocacy coalitions promoting organic and mainstream farming respectively and of fora for discussions between them in the same 18 countries – and the possible correlation with organic farming development. The analysis ends up in an encompassing comparison of the development of institutional interrelationships involving organic farming in six EU member states and its impact on national organic farming growth.

The Impact of Policy Instruments on Organic Farming Growth

The basis for discussing the impact of policy instruments on organic farming growth is Michelsen and Soegaard's (2001) rough comparison of organic farming growth in the 15 EU member states plus Norway, Switzerland and the Czech Republic between 1985 and 1997. The theory behind the study is that the introduction of organic farming should be expected to follow the pattern of the spread of an innovation (see Padel, 2001). This implies that over time the number of organic farms grows in absolute terms, while the rate of growth declines. From this a simple model is developed in which the annual change in the growth rate of organic farms forms the basis for predicting the number of organic farms if organic farming only followed the pattern predicted by innovation theory. By comparing the predicted number of farms with the actual number, differences may appear and the theory is that they should be explained as outcomes of concrete events that have either stopped growth or extended it above prediction. The comparison was done by means of graphs – of which seven are annexed to this article including the figures on the growth of organic farming in the EU as a whole and in the six member states selected for in-depth studies in the discussion below.[3]

Graphical representations of changes in growth rates for organic farms were made for all 18 countries for which information is available in Foster and Lampkin (2000) supplemented with some additional information for previous years.[4] The analysis of the graphs was done by considering for each country as much qualitative information as possible on enduring developments or single events, which were said to have influenced changes in the size of organic farming growth.[5] The introduction of the two common EU regulations on certification and on financial support to organic farming was included in the analysis of all EU member states by the year they were implemented. As the year of implementation differs both between schemes and between countries, there was reasonable space for analysis. All national support schemes were also included in the analysis. Other changes and events were taken into account in a less systematic way. The analysis was done on a purely qualitative basis for all 18 countries. To this was added a rough quantitative analysis based on only eight countries, which were the only ones to include relevant data regarding changes in the growth rate for organic farms.

The general reasoning can be illustrated by the annexed figures. Figure A1 for the EU as a whole shows permanent growth with a tendency towards increasing growth rates rather than the expected decline. Here, the reasoning is that some actions have prevented growth rates from falling, not least during 1992 and 1994 when growth rates were high. The Figure A6 on Italy shows a pattern of endless growth during the 1990s, which – like the pattern for EU as a whole – is expected to imply that several actions positive for organic farming growth have taken place during the period. The Figure A2 of Austria shows growth rates declining to 0 after some growth in the 1980s and a strong impulse in 1992. The development in Denmark (Figure A4) shows strong variation in the growth rate, even involving a period of decline in the number of farms – thus for Denmark it is expected to find actions with a negative impact. Austria, Italy and Denmark have organic farming sectors larger than the EU average. The remaining figures concern countries with small organic farming sectors, but their growth patterns also suggest varying external impulses for organic farming growth.

The main analysis involved a crude test of the rather simplistic assumption that the introduction of policy instruments served as the major impulses for growth in the number of organic farms. Regarding legal instruments in support of organic certification, the full qualitative analysis suggests that the introduction of both national and EU legal instruments had some positive impacts on the rate of farmers' conversion to organic farming. It was, however, difficult to detect impacts that undoubtedly referred to the introduction of a (new) certification system. However, in cases where a uniform national certification system was introduced, only positive correlations with organic farming growth in the following year(s) appeared. In cases of competition between national production standards, negative impacts on growth were found with Germany as the main example. The supplementary quantitative analysis indicated an even stronger conclusion by suggesting a rather strong and statistically significant positive impact of introducing the common EU standards in the first half of the 1990s. Compared to the predicted number of organic farms, both quicker growth (acceleration) and real growth (more farms) were found.

Regarding financial instruments the analysis is limited to include only public support paid to organic farmers. Here, the qualitative analysis points towards a positive impact on the growth of organic farms in absolute terms when economic support was introduced for the first time – whether originating in national or EU support schemes. Subsequent changes, whether in national support or through replacing national support with EU support, seems only to have accelerated the growth process. Examples are the introduction of EU support in Austria in 1995 and the introduction of permanent support in Denmark 1993. In Austria the introduction of EU support was followed by acceleration, in Denmark by stagnation. The general finding was not contradicted by the quantitative analysis. It suggested that both national and EU support had a statistically significant impact on growth, but no effect on the long-term size of the organic farming sector. Hence, the rather provocative conclusion of both the qualitative and the quantitative analysis is that public support paid to farmers may have a clearly positive initial impact while subsequent changes accelerates only organic farming growth leaving the long-term predicted number of organic farmers unchanged.

Finally, an intricate interplay between certification and support was found with certification appearing a necessary precondition for both developing organic farming and introducing financial support. In many countries EU certification was introduced about the same time as EU support to farmers. Hence, in these instances it is difficult to separate the effects of EU certification and support. For the remaining countries, the quantitative analysis suggests, however, that the introduction of EU certification had an absolute impact on growth whereas the impact of EU support was mainly to accelerate the development.

Thus, the rather rough and tentative analysis of growth patterns in 18 European countries suggests that political instruments have influenced the development of organic farming but mainly by initiating a development. On the other hand, the total impact of all instruments seems not very high and policy appears not to have enduring effects on growth. Furthermore, the growth of organic farming seems to depend more on the introduction of common production standards than on support paid to farmers. This finding goes well in hand with the theoretical emphasis put above on the importance of organic farming identity. More pragmatically, the finding suggests that it appears paramount for the ability to attract farmers in the long run that organic farming reaches a uniform and clear definition and that the introduction of EU standards has helped to fulfil this precondition. It adds to the reliability of this result that it was confirmed by national key informants in their response to qualitative questions regarding the contribution of public policy to the general development of organic farming (as part of the response to the questionnaire mentioned in the next section).

Michelsen and Soegaard (2001) also consider the importance of other factors for organic farming growth and conclude that individual factors appeared to have some impact on national growth in each country. Among important factors in at least a few countries is the economic position of organic farmers when compared to non-organic farmers. It seems that in periods of general agriculture recession, farmers may be more inclined to look for supportive arrangements, such as support for organic farming than under prosperity. This factor seems an important explanation for the recent strong growth of organic farming in United Kingdom – and a contributory cause in Switzerland. A second factor contributing to organic farming growth is moral suasion among farmers (a communicative policy instrument), which may see conversion to organic farming as an individual reaction to politicisation of agri-environmental issues. This factor might help explaining why organic farming uptake is relatively high in German speaking and Nordic countries. A third factor found in several countries includes the development of contacts to supermarkets in the food market and of other institutional networks.

The Presence of Conditions for Policy-Oriented Learning in Organic Farming Policy

The issue of this part of the empirical analysis is to compare whether the extent to which elements of advocacy coalitions and other preconditions for policy oriented learning are found varies with the size of the organic farming sector. The source for the analysis is responses received in 1997–1998 to a questionnaire sent to key informants in the 18 countries mentioned above.[6] The main issue of the questionnaire was to identify different types of actors and alliances within the field of organisations and politics in each country and their interrelationships with regard to policies in support of organic farming. Hence, the issue of belief systems and values within different alliances and advocacy coalitions is presupposed to follow the distinction between organic and mainstream agriculture and not addressed directly.

One of the main issues mentioned in the theoretical considerations is organic farming identity, which is important for the ability to establish own institutions and to act within the general institutional environment of agriculture. One measure of organic farming identity is the presence of organisations representing organic farming. In the questionnaires between two and eleven organisations were mentioned among the most representative organisations of organic farming in each country. In all countries, private organic farming organisations were included among the three considered most representative for organic farming. However, only in three countries (Germany, Belgium and the Netherlands) a private organic farming umbrella organisation is mentioned as the most representative one. In nine countries, it was one among more competing organic farming organisations that was perceived the most representative for organic farming. In the remaining four countries the most representative organisations were either research and development organisations (Norway and Switzerland – both countries with quite high shares of organic farming), a commercial certifying body (Greece) or – as in Spain – a shared function involving both organic farming organisations and governmental agencies. Hence, organic farming has a clear identity within private organisations in all countries although in a few countries the identity is combined with some expertise of science or certification. In responses received to another question it is stated that cooperation between organic farming organisations is found in all countries. However, in eight countries competition between organic farming organisations is also found and this emphasises that the plural identity of organic farming with regard to organisation may also leave some confusion with regard to representation.

Another aspect of organic farming identity is the extent to which organic farmers are members of general farmers' organisations. Organic farmers are found as members in at least one general farmers' organisation in all countries except Luxembourg. In Austria, membership of one distinct agriculture organisation is even obligatory to all farmers – whether organic or not. The level of integration of organic farmers into the general agriculture organisations varies much between countries. In Greece and Italy, organic farming is but a small activity of the general organisations, while in Denmark and Switzerland cooperation is so strong that organic farmers are represented in the boards of (some of) the general organisations. Hence, on the level of farmers, the distinction between organic farming and mainstream agriculture institutions is not clear-cut. When considering the level of organisations, cooperation between organic farming and some general farmers' organisations are found in all countries, while examples of open non-cooperation (that involves a potential for competition) are mentioned in eleven countries. In sum, the interrelationships between organic farming and general farmers' organisations are mixed.

Whereas general farmers' organisations from the outset are considered members of a competing advocacy coalition, environmental organisations appear potential members of an organic farming advocacy coalition. This seems also to be the case in most countries, as it was only in Greece, Spain and the U.K. (all with small organic farming sectors) that environmental organisations had not made active positions in support of organic farming. In most other countries – irrespective of the size of the organic farming sector – organic farming is supported because of its positive impacts on the environment, while in Luxembourg, Germany, Switzerland and Finland the issue of biodiversity seems important. It is, however, only in Denmark and Switzerland that responses mention that environmental organisations have incorporated organic farming in their political strategy and are aiming for influence on the organic farming production standards and practices. To sum up, the environmental organisations are supportive in general to organic farming but they seem not very active in organic farming advocacy coalitions.

When asking for the active alliances in organic farming policy-making, the type of formalised alliance mentioned most frequently – in eight countries – includes only organic farming organisations and public agencies. In four of these countries an additional alliance is mentioned, which also include general agriculture institutions. These findings indicate that organic farming policy to some extent is separated from general agriculture policy. When considering the other alliances mentioned, the impression of separation is emphasised, because organic farming and general farming organisations themselves only constitute alliances in two countries. Other alliances include organic farming organisations only (two countries) or in cooperation with either political parties (six countries) or research and development institutions (four countries – all of which have also alliances with public agencies).

The impression of a separate policy subsystem for organic farming is confirmed by the responses given to a question on the general level of conflicts with regard to political decisions on organic farming. Here it is clear, that discussions on pro's and con's with regard to support for organic farming is an issue in all countries. In six countries with large organic farming sectors, however, the level of conflict is considered low or medium, while in countries with comparatively small organic farming sectors the level of conflict is characterised as medium to high. Hence, the discussion is more heated where applications for support are less frequent. The contents of the conflicts mainly concern the level of support. Stronger and more varied conflicts are mentioned with regard to administrative matters. In Italy the level of conflict regarding administration is even characterised as "very high", while in seven countries (with different sizes of the organic farming sector) the level of conflict is characterised as "high". The emphasis put on administrative matters reflects two different issues. One issue (mentioned by the Italian respondent and others) is that public agencies and general lobbying organisations do not find organic farming an interesting issue – hence here is an issue of making organic farming visible within the administrative system. Another issue is unclear responsibilities for different agencies within the public sector (mentioned by respondents from Switzerland, Denmark, Belgium and the Czech Republic) – i.e. matters of an emerging subsystem of organic farming bureaucracy as part of a policy subsystem.

The questionnaire also covered the issue of fora for discussing problems of organic farming policy. Issues of certification appeared to be resolved by different types of fora in twelve countries, while issues of conversion aid and related matters are discussed in fora in nine countries and other types of support are discussed in different fora in up to eight countries. Sweden and Finland (with large organic farming sectors) have no fora for discussion at all, while Greece is the only other country without a forum for discussing certification. There are quite different conditions for discussion. Most fora for discussion are organised within public agencies, while in some countries, discussions are organised in councils involving private actors. It is only in the latter case that the fora may represent a place for exchange of views relevant for policy-oriented learning. Furthermore, although certification is an important matter for organic farming identity, it may be too narrow to attach interest among others – and it may even be a matter of dispute whether other than organic farming organisations should be included in this kind of discussion. Hence the most important discussions in a political and organisational context – those regarding conversion and other issues – have only few fora and most of them are not organised in a way that attracts participation from other interests than those of organic farming.

Finally, the issue of influence from different type of actors on the different phases of the organic farming policy process was evaluated in the questionnaire. Two types of actors were distinguished: those of the public sector and those of the private sector including non-profit organisations of the civil society. Both types of organisations were perceived to have substantial influence in at least one of the three phases: policy formulation, decision and implementation in all countries. On the average, private organisations were perceived to have "medium high" influence on policy formulation with public organisations slightly below. Regarding both decision and implementation, the influence of public sector organisations was perceived as "medium high" while the influence of private organisations was considered lower – especially regarding decision. Among countries with large organic farming sectors, public organisations are in general seen as very influential in all phases of the policy process. The two main exceptions from this pattern are Denmark and Switzerland and in both countries private organisations are considered more influential than public organisations in all policy phases. Among other countries, the level of influence to both types of organisations are in general considered lower with France as the main exception characterised by equally strong influence in all phases attached to both public and private organisations.

The answers obtained from this survey should not be taken for more than a first and very rough attempt to address the issue of organic farming policy making on a comparative basis. The findings suggest that there is no clear cut division between organic farming and mainstream agriculture organisations neither on the level of individual farmers nor on the organisational level and that organic farming's relationship with environmental organisations is not so strong and friendly as one might have expected. The alliances in organic farming policy have a heavy load on organic farming organisations and public agencies, while general farming organisations seem rather less important. This turns the attention to the fact that organic farming policy may be developing as a policy subsystem separate from the mainstream agriculture policy subsystem – a suggestion that is strengthened by the fact that political conflicts are perceived higher regarding administrative matters than regarding acceptance of support paid to organic farmers. The impression of the development of an organic farming policy subsystem seems also confirmed by the presence of rather specific fora for conflict resolution. Both the emphasis put on certification rather than issues of general organic farming development within agriculture and the emphasis put on public agencies rather than councils including private interests signify a low priority attached to fora for discussion between organic farming and general agriculture institutions – i.e. for a factor that in theory should facilitate a process of policy-oriented learning and change. Finally, the picture of the emergence of a separate policy subsystem seems confirmed by the high level of influence in all phases of organic farming policy development attached to private and public organisations – not least in the countries with a large organic farming sector.

Regarding the possible correlation between the presence of organisational conditions for policy learning and organic farming sector size, the material shows no clear pattern connecting countries with large organic farming sectors with distinct characteristics on the above mentioned aspects of the theory of advocacy coalitions. One strategy for further analysis might be to see the differences in organisation of organic farming policy as a matter of policy styles, where each country has special characteristics (Howlett, 1991 – see Mol et al., 2000 for a similar recognition with regard to the importance of national policy). However, the strategy followed here starts from the fact that the survey covered only the presence of preconditions for policy change through policy learning while the theoretical considerations emphasised the qualitative character of interrelations between organic and mainstream farming institutions. Thus, the last empirical analysis attempts to analyse the qualities of institutional interrelationships.

Institutional Interrelationships and Organic Farming Growth in Six Countries

The aim of this part of the empirical analysis is to discuss whether learning processes have taken place between advocacy coalitions in a broader context than policy i.e. including the civil society and the market. It is done on the basis of a comparative study of six of the 18 countries analysed above, focussing on interrelationships within the four domains of agriculture's institutional environment: agriculture policy, the farming community, the food market and an institutional setting capable of moving across the other domains (Michelsen et al., 2001). The six countries include Austria, Denmark and Italy representing countries with large organic farming sectors and the U.K., Belgium and Greece representing countries with small sectors. In five countries the situation of the whole country was covered while in Italy the case was made on the basis of in-depth studies of two strongly diverging regions (Marche and Sicily) to exemplify the large variation between Italian regions. The figures in the annex show that development patterns vary both between large-sector-countries and between small-sector-countries.

The methodology followed is purely qualitative. The empirical basis consists of systematically collected assessments from local actors and observers in addition to those used in the above analysis, but this time institutional interrelationships within each domain of the agriculture institutional environment was the theme of the study. The analysis is thus focused on describing institutional interrelationships within each of the six countries by means of the concepts of pure competition, pure cooperation and creative conflict and see whether the theory is right in emphasising creative conflict as more fertile than the others for organic farming growth.

First of all, it appears from the six country studies that a varied institutional approach is justified partly by confirming the findings of the analyses above and partly by adding new arguments. The presence of subsidies paid to farmers appeared important for farmers' propensity to convert to organic farming – but only in some countries and not necessarily those with the largest organic farming sectors. Relatively large uptakes motivated by subsidies were found in Austria and Greece and in individual regions of Belgium and Italy. However, in Denmark and other regions of Belgium and Italy, similar subsidies did not trigger much conversion. In the food market domain, consumer demand for organic products has consistently been high in Denmark, Belgium and the U.K., but this did not in itself trigger increases in the number of farms – hence there seem to be institutional barriers for transformation of demand into supply. Finally, an institutional setting for cooperation between organic farming and general agriculture institutions plays an important role in the Danish organic farming development, while it is lacking completely in the U.K. and Greece and weak in the remaining countries.

While policy change may be quite easy to identify in terms of laws or other political decisions, changes in other domains are far less clear and hence more difficult to identify and analyse while applying a policy learning perspective. The method used in this study is to look for institutional change within each domain, using *organisational changes* as indicator. Institutional change is thus seen as a manifestation of the working of the dynamics of institutional interrelationships. Hence, the theoretical model is to explain organic farming growth by the dynamics of institutional interrelationships, which manifest themselves in changes of organisation within each domain. Not only the number of organisational changes is relevant as a measure of institutional change – also the range of institutional change need to be taken into account in terms of the number of domains involved.

The results of the empirical analysis of the six countries are summarised in Table 2. It shows firstly, that only the two countries with the largest organic farming sectors have experienced institutional changes, which involved all four domains. Institutional changes in countries with small organic farming sectors involve only changes in two or three domains. Secondly, the table includes a summary for all six countries of the institutional interrelationships between organic farming and mainstream agriculture institutions, which are suggested to explain the institutional changes. It is noteworthy that no interrelationship within any domain in any country is characterised by either *pure* co-operation or *pure* competition. Within all domains interrelationships were characterised by *leaning* either to one of the extremes or to creative conflict. Only when adding up characteristics for the interrelationships within a country, the extreme categories of pure cooperation/competition appeared relevant.

The table shows that the overall characterisation of pure cooperation is found in the two countries with the largest and the smallest organic farming sector respectively, while the two other countries with a small organic farming sector are characterised by pure competition. The remaining two countries with large organic farming sectors are characterised by a mix of cooperation and competition. Denmark is characterised by creative conflict, while the mixed situation in Italy is caused by different developments in different regions. Hence, the overall situation is not so clear as one might have expected from the theoretical considerations: at first sight, creative conflict seems not the type of interrelationship that most effectively promotes organic farming growth. There are, however, concrete factors that help explaining the deviations from theoretical expectations.

First, regarding Austria, organic farming was in fact introduced as a strategy for developing all Austrian agriculture at the time when the negative impacts of future EU membership for national support of agriculture was considered in the late 1980s and early 1990s – not least expressed by the late minister of agriculture, Riegler. This accords well with the exception mentioned under the theoretical considerations regarding pure cooperation and helps explaining the extensive institutional changes in Austria. Stagnation in the development of Austrian organic farming set in when EU membership was realised in 1995 as it appeared that the EU membership made other solutions available to Austrian farmers. When realising the strong growth in Greece associated with pure cooperation and this is combined with a similar tendency in Sicily, it seems – keeping the Austrian experience in mind – that pure cooperation allows for more organic farming growth than pure competition. In addition, when considering the ongoing stagnation in Austria it seems that pure cooperation allows only organic farming growth to a certain extent. The limit seems to be

Table 2. The Interrelationship Between Organic Farming and General Agriculture Institutions in Six Countries by Societal Domain.

Variables/ Domain	Austria	Denmark	Italy	United Kingdom	Belgium	Greece
Organic farming sector size[1]	9.6%	3.5%	1.8%	0.6%	0.6%	0.5%
Number/ extent of institutional changes[2]	2 4 domains	3 4 domains	2 3 domains	2 3 domains	2 3 domains	1 2 domains
Overall assessment	Pure co-operation	Creative conflict	Competition/ co-operation	Pure competition	Pure competition	Pure co-operation
Farming community	Co-operation	Creative conflict	Competition/ co-operation	Competition	Competition	Co-operation
Agriculture policy	Co-operation	Creative conflict	Co-operation – weak	Co-operation – weak	Competition	Co-operation
Food market	Creative conflict developing	Creative conflict	Competition/ emerging co-operation	Competition/ emerging co-operation	Competition	Co-operation/ exports
Institutional setting	Co-operation. Dominated by general farming institutions	Creative conflict. Dominated by organic farming views	Co-operation – no impact on development	Missing	Competition – involves only organic farming organisations	Missing

[1] Organic share of the total number of farms.
[2] Extent of institutional changes includes the maximum number of domains involved in any change.
Source: Michelsen et al. (2001): Chapter 8.

when other options appear equally economically attractive. Within the theoretical framework a probable explanation is that under conditions of pure cooperation re-conversion to non-organic farming appears less burdensome than under conditions involving conflict or competition.

The Italian case falls outside the possibilities mentioned theoretically. The Italian situation of endless growth seems the result of a situation where organic farming was developed early in northern/central regions under conditions of competition. When it stopped there, organic farming growth took off in southern regions and islands under conditions of cooperation, and recently a domestic market of high prices in the far northern regions seems able to absorb domestic production. In Italy a sequence of development in different regions have thus – more or less by accident – established a situation that bears some similarity with creative conflict.

When moving into the distinct domains of the institutional environment, it appears that interrelationships within the domains of the farming community and of the institutional setting are more important than interrelationships within other domains. Competition between organisations of mainstream and organic farming in the farming community seems to hamper growth as exemplified by Belgium and the U.K. – and by the northern/central regions of Italy. This seems quite understandable as organic farmers are recruited among farmers with a perception of agriculture, which is strongly reflected in or influenced by mainstream farming organisations. However, the distinct identity of organic farming is important for its survival and therefore cooperation only seems practicable to the extent that organic farming's identity does not wither away. Regarding the institutional setting the main thing appears to be to have one – and when it is there, that both sides are combined and that the organisations of the institutional setting are issued with some influence on development.

Agriculture policy may lead to organic farming growth under conditions of cooperation, but once again it seems important that cooperation does not lead to a complete silence regarding differences – not least because policy under conditions of weak cooperation only appears to have limited impact on growth. Finally, with regard to the food market, the table should be read with caution because only little information was available for most of the countries. However, creative conflict is found in the two countries with the largest organic farming sectors,[7] while competition reigns in three of the four other countries – and in the fifth country – Greece – domestic production is not used as a basis for developing a domestic market for organic food.

To sum up the findings of the broad institutional comparison of organic farming development in six countries, it offers a rather clear explanation for the rather unclear findings in the first two empirical analyses of organic farming development. Organic farming development depends to a major extent on the local conditions for interrelating with mainstream farming organisations. Hence, farmers do not take up a policy instrument, such as subsidies, if agriculture policy does not fit into institutional changes taking place in other domains. As expected in theory, the empirical analysis suggests that organic farming has many difficulties in developing under conditions of pure competition. Purely cooperative interrelationships may help promoting organic farming to reach a certain level rather quickly (as seen in Austria and Greece), but when other options appear available to farmers, then farmers seem less prepared to stick to the values of organic farming than under less cooperative conditions where the organic farming identity is more distinct. This is clearly exemplified by the current stagnation in Austria. In this perspective, more sustainable organic farming growth may be reached under conditions of creative conflict, which implies that organic farming identity is not under threat from being silenced out (as under pure cooperation) or suppressed (as under pure cooperation) and where positions in all domains have to be fought for within a trial-and-error process.

CONCLUSIONS AND DISCUSSION

The aim of this article was to cope with the major variations in the growth of organic farming observed in European countries, in spite of a relatively common regulatory framework, by seeking to pick up contributions from empirical analyses in support of either of two lines of argument. One line of argument states that policy initiatives are effective and the causes for organic farming growth. The other line stresses the attractiveness of organic farming for farmers as a full production system that internalise environmental and other politically salient concerns.

The major reasons for opposing the two lines of argument is: (i) that organic farming has developed outside and in overt opposition to the values of mainstream agriculture; and (ii) that agriculture usually is seen as a political subsystem left to self-rule i.e. by the views maintained within mainstream agriculture. The theoretical tools for analysing this issue include first a specification of policy instruments, which may be used to intervene into or support a specific development in society. They include legal, financial and communicative instruments, which are based on the rationales of the state, the market and the civil society respectively and are all relevant in support of organic farming although it originally developed as self-regulation within the civil society. The second theoretical tool includes the notions of advocacy coalitions and policy learning between organic and mainstream farming coalitions as means for grasping policy change within an institutional framework. Finally, a specification of three distinct types of interrelationship that may develop between adverse positions such as organic and mainstream farming was developed, i.e. pure cooperation, pure competition and creative conflict each of which were expected to have different impacts on organic farming growth. The analyses done with these tools were not limited to the policy domain alone because agriculture as an industry still must rely to some extent on the market and because agriculture's organisations within the farming community are important carriers of agriculture' self rule as is an institutional setting capable of combining efforts across domains.

Each of the three empirical analyses is based on the best information available, which, however, draws strongly on qualitative assessments done by local scholars and actors. Hence, the analyses can only represent a first approach to the issues under consideration. The first empirical analysis concerns a rather simplistic version of the first line of argument namely that policy instruments in support of organic farming certification and of organic farmers was the main trigger of organic farming growth. By comparing the development in 18 European countries – including EU member states as well as non-member states – some impacts appeared, *viz.* the importance of a common legal basis for certification, while the importance of financial support was toned down. To sum up, policy had some impact on growth, but no clear explanation for the variation in organic farming growth was obtained from the analysis.

The second empirical discussion suggested that organic farming had been able to establish advocacy coalitions and join processes of policy learning together with mainstream agriculture institutions. However, when analysing 17 of the 18 countries from the first analysis, the distinction between organic and mainstream farmers appeared unclear both on the level of farmers and on the organisational level. Furthermore, it was difficult to find clear actors outside agriculture that supported organic farming. Hence, the main impression from the analysis was that a separate policy subsystem for organic farming has developed in most countries and that there was no clear contribution from this type of analysis to explaining differences in organic sector size.

The third empirical discussion was based on the concepts of institutional interrelationships. In this analysis only six countries were included – representing as much diversity among EU member states as possible. The analysis suggested that the inclusion of local conditions is very important when attempting to explain the impacts of a common policy. This general statement was specified by realising that organic farming growth depends on interplays between a plurality of domains – the more domains involved the better – and that some kind of interrelationship other than pure competition between organic farming and mainstream agriculture organisations is necessary for organic farming growth. Interrelationships within the farming community and the institutional setting had special significance.

When allowing for the slack in information, the analyses taken together suggest that one should not rely too much on promoting organic farming only by means of policy instruments. They clearly had an impact as signals of official recognition of organic farming, but they were far from decisive for the development. Furthermore, the impact of the attempt to promote organic farming by political decisions was limited by the fact that rather than developing advocacy coalitions within the national subsystems of agriculture policy it seems that separate policy subsystems for organic farming are developing in most countries. The best suggestion (which needs to be confirmed by further research) for explaining organic farming's remarkable status in some EU member states points to the importance of the nature of local interrelationships under which organic farming has developed. Conditions of competitive interrelationships between mainstream agriculture and organic farming hampers the development, while cooperative interrelationships promote organic farming development although the organic sector seems vulnerable to changes in perceptions of the interests of mainstream agriculture. Finally, creative conflicts seem to be the kind of interrelationship that form the basis for a more enduring growth of the organic farming sector on the basis of trial-and-error processes within all domains.

The general implication of these findings are that organic farming must be able to establish and sustain a clear identity and appear able to act autonomously in all domains. This points indirectly to the fact that organic farming does not seem attractive for farmers mainly or only as a means for obtaining incomes from policy support or price premiums in the market. The ability of organic farming organisations to cope with issues in all domains of society in the countries with many organic farmers suggests that it is important for growth that organic farmers support the values behind the production system – and are prepared to and allowed by the political system to keep up elements of self-regulation. In this way the overall conclusion is that policy in support of organic farming matters, but much more so when it is supported by a genuine interest among farmers in choosing production system that internalise environmental and other concerns such as organic farming. Hence, the main conclusion of this investigation is that future and more systematic research should head for explanations of organic farming growth in Europe that combine the two lines of argument.

NOTES

1. One major initiative includes a series of volumes from a project on economic and political aspects of organic farming development in Europe issued from University of Hohenheim, Stuttgart. Further information is available at http://www.uni-hohenheim.de/~i410a/ofeurope/. Another initiative is included in *Sociologia Ruralis* vol. 41/1 – a special issue on organic farming.
2. "Effects of the CAP-reform and possible further development on organic farming in EU", sponsored by the EU Commission under the Fourth Framework Programme: FAIR3-CT96-1794.
3. It should be noted that the axes in the figures are based on logarithmic scales to level tendencies to exponential growth.
4. See, http://www.organic.aber.ac.uk/stats.shtml
5. Most of the qualitative information of events with a potential impact on organic farming development in each country was collected for other purposes in the project while other information was generously made available to the author by dr. Nic Lampkin and Susanne Padel of the Welsh Institute of Rural Studies, University of Wales, Aberystwyth.
6. The author has designed the questionnaire. Responses were received from 17 countries, Portugal being the only country missing. The responses were based on investigations made by the key informants in each country including interviews with key actors. Hence, the basis for the following discussion is highly qualitative assessments made by informed students of each country.
7. Lynggaard (2001a) and Michelsen et al. (2001a) include descriptions of the development of the Danish organic food market development, which illustrates the meaning of creative conflict very vividly.

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ANNEX. VARIATION IN GROWTH RATES FOR THE NUMBER OF ORGANIC FARMS 1985–1997 IN EU AND SIX SELECTED MEMBER STATES

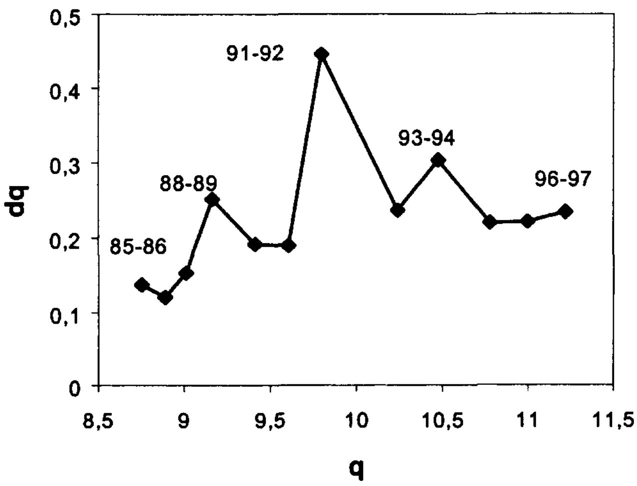

Fig. A1. Changes in the number of organic farms 1985-1997 in 15 EU countries. Growth rates (dq) and total number of organic and certified and in-conversion farms (q), logarithmic scales (log n).

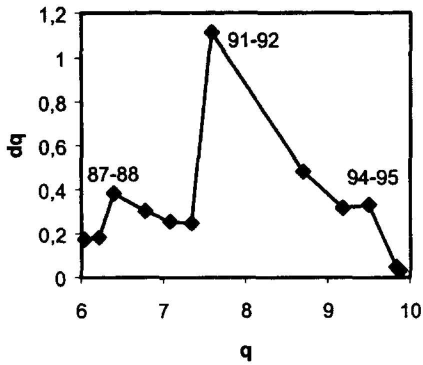

Fig. A2. Changes in the number of organic farms 1985-1997 in Austria. Growth rates (dq) and number of farms (q)), logarithmic scales (log n).

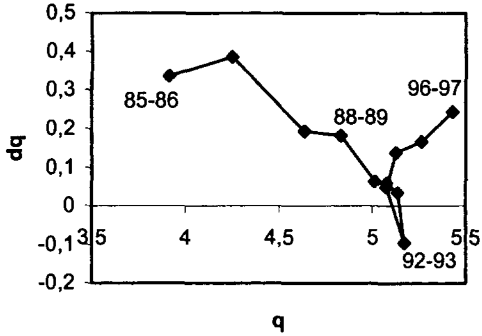

Fig. A3. Changes in the number of organic farms 1985-1997 in Belgium. Growth rates (dq) and number of farms (q), logarithmic scales (log n).

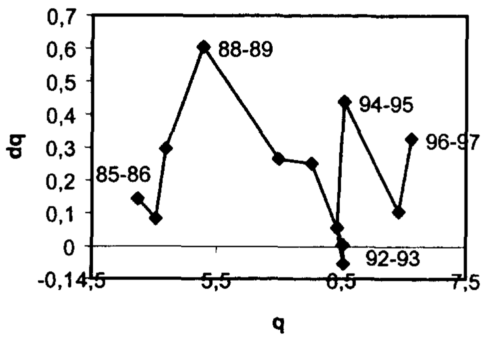

Fig. A4. Changes in the number of organic farms 1985-1997 in Denmark. Growth rates (dq) and number of farms (q), logarithmic scales (log n).

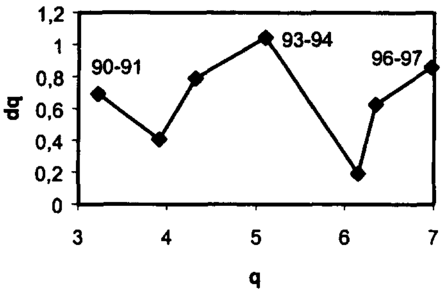

Fig. A5. Changes in the number of organic farms 1985-1997 in Greece. Growth rates (dq) and number of farms (q), logarithmic scales (log n).

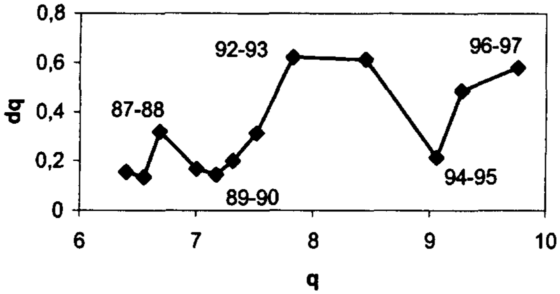

Fig. A6. Changes in the number of organic farms 1985-1997 in Italy. Growth rates (dq) and number of farms (q), logarithmic scales (log n).

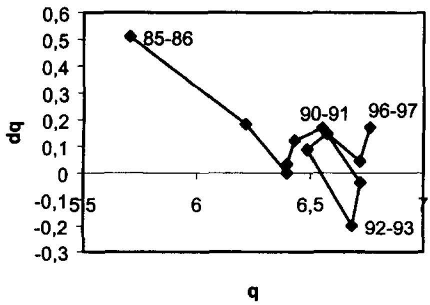

Fig. A7. Changes in the number of organic farms 1985-1997 in the United Kingdom. Growth rates (dq) and number of farms (q), logarithmic scales (log n).

7. ENTRY AND EXIT IN CALIFORNIA'S ORGANIC FARMING SECTOR

Karen Klonsky and Martin D. Smith

ABSTRACT

In California, organic acreage increased by 60% and sales of organic commodities increased by 110% between 1992 and 1997. The rate of growth in the organic industry does not reveal the dynamic nature of California's organic agriculture. In this chapter, we explore the characteristics of farmers entering and exiting the organic market in California. In so doing, our analysis provides insight into the impact of policy and growth on the future composition of the organic industry.

I. INTRODUCTION

Organic agriculture has attracted many farmers by allowing them to forego the use of synthetic pesticides and fertilizers and in some cases lower costs of production. At the same time, increased consumer demand for organic products has fueled a rapid expansion of organic production in the United States. The high prices commanded by organic products and often unmet demand for organic products have enticed growers to enter into the organic market. However, the transition to organic production practices has its challenges as does the particularities of selling into the organic marketplace. The learning curve is steep and it may take several years for a farm's unique agroecosystem

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to transition fully to a functioning organic system. Organic markets also tend to be volatile and easily saturated. Not surprisingly, in any given year there are growers entering the organic subsector of U.S. agriculture while others leave.

The burgeoning domestic and international organic market led the organic farming community to recognize the need for standardized labeling of organic commodities. While it is not without controversy, the basic thinking is that standardization will facilitate trade and bolster consumer confidence. The industry's interest in standards eventually prompted state and federal regulators to act. The long road to federal regulation of organic labeling has not been without bumps and detours. There is concern within the industry that the new regulations could act as a barrier to entry for small growers, and that this barrier would pave the way for large conventional growers to enter the organic market and displace smaller existing organic growers. In this chapter we present a picture of organic agriculture in California in the context of U.S. production and the evolving regulatory environment for organic agriculture.

Organic cropland acreage in the United States more than doubled between 1992 and 1997 reaching 850,000 acres (Anton Dunn, 1997; Green, 2000). Although this represents only one quarter of a percent of total cropland acres, organic agriculture provides an important market niche for many growers. Adoption of organic agriculture varies by commodity, the most prevalent being fruit, vegetables, and specialty grain sectors. For example, nationwide in 1997 only one-tenth of a percent of corn acreage was under organic cultivation while 2% of apple, grape, lettuce, and carrot acreage was organic (Green, 2000). California is second only to Idaho in total organic crop acreage. Idaho is dominated by field crops, while produce (vegetables, fruits, and nuts) comprises the majority of California production. In fact, California accounted for almost half of the organic vegetables grown in the United States in 1997.

Organic acreage in California increased by 60% and sales of organic commodities increased by 110% between 1992 and 1997 (Tourte & Klonsky, 1998; Klonsky et al., forthcoming). In sharp contrast, the number of growers increased by only 32%. The rate of growth in the organic industry be it measured by acreage, sales, or number of growers, while important in and of itself, does not reveal the dynamic nature of California's organic agriculture. In any given year, over a fifth of the growers are new to the program and slightly under a fifth leave the program. Thus, the steady growth in California organics is comprised of many new entrants each year offset partially by many exiters from organic farming. Looking only at aggregate annual growth rates masks this rate of turnover. The entry of relatively large conventional growers into organic production coupled with the new federal policy for organic commodities creates uncertainty about how the new organic agriculture will look.

In this chapter, we explore the characteristics of farmers entering and exiting the organic market in California. Previous studies have focused on describing the size and growth of the organic market but have not addressed the substantial turnover of farmers within the industry. In so doing, our analysis provides insight into the impact of policy and growth on the future composition of the organic industry. Section II provides further background on organic agriculture in California and describes the legal requirements for registration in California and certification of all organic farmers. Section III describes the data used in the models. Section IV outlines the statistical model of exit decisions. The results of the estimation are presented and briefly discussed. Section V uses several statistical tests to compare the characteristics of entrants and incumbents in the organic market. Section VI discusses the results of the previous sections and draws implications for the future of organic agriculture in California.

II. BACKGROUND

California organic agriculture expanded rapidly from 1992 to 1997, with double digit growth in registered acreage and sales (Tourte & Klonsky, 1998; Klonsky et al., forthcoming). Growth in these measures was considerably faster than in California agriculture as a whole, although organic farms accounted for 0.8% of all sales or less for each commodity represented, excluding livestock and products, and 1% for produce alone (CDFA, 1999). Organic vegetable, fruits, and nut crops (produce crops) have made the greatest inroads in the organic subsector of California agriculture. Together they represented 89% of farms, 74% of acreage, and 90% of revenue for organic production in 1997 (Table 1). This is in striking contrast to California agriculture as a whole where 38% of harvested cropland was devoted to produce that contributed 47% of gross revenue in 1997 (CDFA, 1999). Field crops accounted for 26% of acreage and 6% of organic sales in 1997 compared to 62% of all harvested acreage and 13% of gross revenue for all of California agriculture. The most important difference between the composition of organic agriculture and the entirety of California agriculture is in livestock, poultry, and products,[1] which represented only 2% of gross sales in the organic subsector but 25% overall.

In 1997 the median income for organic farms was $8,000 (Klonsky et al., forthcoming) and only 25% grossed over $40,000. The top 10% grossed over $177,000 and realized 75% of the sales. The median farm size was only five acres while only 10% of farms were over 75 acres. Not surprisingly, farm size tended to vary with commodities grown. Farms growing field crops tended to be larger than average with a median size of 55 acres, comprising 10% of farms

Table 1. Number of Registered Organic Growers by Region and Commodity Group as Reported to CDFA, 1997.

Region	Produce						
	Fruit & Nut Crops	Vegetable Crops	Unclassified Produce	Field Crops	Nursery, Forestry & Flowers	Livestock[a]	Total Growers[b]
Bay Area	11	30	4	–[c]	7	–	46
Cascade-Sierra	45	33	11	3	6		85
Central Coast	91	97	9	16	20		197
North Coast	163	143	13	6	20	4	293
Sacramento Valley	104	56	11	53	7	–	194
San Joaquin Valley	100	38	–	10	–		136
South Coast	432	57	14	4	6	3	485
Southeast Interior	73	25	–	4			97
Total Growers	1,019	479	65	97	68	11	1,533

[a] Includes poultry and related products.
[b] Row totals do not add because of multiple responses to commodity group category.
[c] – Data not reported to protect grower confidentiality.

over 500 acres. These numbers reflect the higher level of mechanization needed for field crop operations compared to fruit, nut, or vegetable farms.

From 1992 to 1997 the number of registered organic growers increased by 369 (Table 2). However, looking at the exit and entry into the ranks of organic growers reveals that an average of 341 new growers registered every year and an average of 267 growers failed to renew their registrations, either leaving the organic market or giving up farming altogether. As such, the sector is characterized by steady growth and a high turnover in growers.

Regulation of Organic Production

The California Organic Foods Act (COFA), signed into law in 1990, protects producers, processors, handlers, and consumers by creating legal recourse for fraud, assuring that foods produced and marketed as organic meet specified standards. COFA calls on the state of California to develop these standards and regulate the production, processing, handling, and labeling of organic products. As part of the regulatory process, COFA requires annual registration of all processors, growers, and handlers of commodities labeled as organic. For growers and handlers, local county agricultural commissioner's offices administer initial registration. Registration information is forwarded to the California Department of Food and Agriculture (CDFA) Organic Program in Sacramento, from which subsequent yearly renewals are generated.

At the national level, the Organic Foods Production Act (OFPA) of 1990 requires USDA to develop national organic standards for organically produced agricultural products and to develop an organic certification program. Devising a set of standards proved more difficult than anticipated, and the initial proposal for the federal rule was not published until December 1997. Over 275,000 comments were sent to USDA, the largest number ever received by USDA (Green, 2000). USDA immediately revised the rule in response to the comments. The National Organic Standards Final Rule for implementation of the OFPA were published in the Federal Register in December 2000 and will be fully implemented in October 2002. The Final Rule includes a list of allowed synthetic and prohibited non-synthetic materials as well as labeling requirements.

The Final Rule requires all growers grossing \$5,000 or more to obtain certification from a USDA accredited third-party certification organization. Organic certification means that a farmer must have an organic farm plan, maintain a paper trail for materials applied, and undergo an annual farm inspection by the certifier. This requirement will affect organic growers who are not certified and are seeking certification for the first time as well as new

Table 2. Entry and Exit Patterns for Registered Organic Growers 1992–1997.

Year	1992	1993	1994	1995	1996	1997
Registered organic growers:						
Continuing from previous year	na	796	1,024	1,090	1,124	1,184
Entering	1,157	333	348	335	345	342
Total number of organic growers	1,157	1,129	1,372	1,425	1,469	1,526
Exiting at the end of the year	361	105	282	301	285	na
Continuing into the next year	796	1,024	1,090	1,124	1,184	na

entrants into organic production. The minimum cost of certification starts at about $200 and can be as high as $2,000 depending on the certifier selected and the complexity of the operation. Several states have state-run certification programs that do not recover the state's full costs and tend to be less expensive than private certifiers.

In recognition of the hardship that certification could cause small growers, USDA made $1 million available in 2001 to share the cost of organic certification in 15 states (USDA, AMS, 2001).[2] Eligible growers must have already been certified in 2000 and could receive reimbursement of up to 70% of the cost of certification with a maximum payment of $500. While this program provides some assistance, it is only temporary funding and available to only a subset of growers.

In order to receive accreditation, a certifying agent must be staffed by personnel with expertise in organic farming practices and be capable of conducting on-site inspections and review certification documents to ultimately make recommendations about certification. The Final Rule prohibits certifying agents from "giving advice or providing consultancy services, to certification applicants or certified operations, for overcoming identified barriers to certification," to prevent conflicts of interest (USDA, 2001). This does not preclude providing information to clients in the form of educational workshops, in-house publications, workshops, or field days. The provision leaves a lot of room for interpretation in terms of what information a certifier can and cannot provide. There is a fine line between education and consulting in many cases. A strict interpretation may leave growers in need of hiring private consultants to help them meet compliance, which would be an additional expense and a possible barrier to adoption of organic production practices.

Arguably, the single most important difference between the federal law and the California law is the certification requirement. While requiring registration of all growers, COFA does not require certification of any growers, although it allows for voluntary certification. It is important to realize that state registration is separate from, and does not act as a substitute for, organic certification. Once the federal law is implemented in 2002, it will supersede the California law, and all organic growers in California grossing over $5,000 will be required to be certified.

Certified Farms in California

Certified growers comprised about 40% of all registered organic growers in any given year. Farms with higher gross sales were more likely to be certified, with 100% of the farms grossing over $1 million certified, but only 30% of farms grossing between $5,000 and $10,000 holding some form of third-party certification. Certified farms, though representing less than half of registered farms, accounted for more than 80% of the registered acreage and about 90% of the registered sales in the state. Several explanations are possible. First, small farmers may consider the increase in sales gained from certification, if any, insufficient to justify the cost to certify. Second, incentives for certification may be greater for larger scale operators who are less likely to direct market their products and more likely to sell to processors or wholesalers, who usually require certification. Finally, certification itself may improve the prospects of a farm through greater market recognition, greater access to markets, or enhancement of production and marketing skills through information exchange among farmers, agencies, and other partnerships.

III. DATA

Data for this analysis were obtained from information provided in the annual registration forms of organic growers registering in the California Organic Program. Registration is required by COFA of any farm entity operating within California that makes the claim of organically produced products in marketing. The data include farmers operating in California marketing products as organically produced regardless of whether the products are sold within California, out of state, or out of the country. The data do not include organic production outside of California that is marketed within California. More importantly, the data do not capture farms in California using organic methods that do not market their products as organic. In general, a farm that is growing under contract to another farm will hold its own registration. There are cases in which the contracting entity is in control of production decisions and facilities. In these cases the larger farm may hold one registration jointly for itself and for the smaller farms with which it subcontracts. The data begin in 1992, the first year of registration in California, and continue through 1997.

The characteristics of California organic growers contained in the registration forms include commodities grown, geographic location, gross sales from organic commodities, and producing acres. In addition, the registrant includes the name of all third-party certification organizations that have certified production in that year.

Individual commodities were classified into six major commodity groups: field crops; fruit and nut crops; livestock, poultry and products; nursery, flowers and forestry; vegetable crops; and unclassified produce (Table 1). These same principal commodity groups are utilized by CDFA in reporting the state's agricultural production (CDFA, 1999). The unclassified produce category was necessary because a number of growers reported production under the broad heading "fruits, nuts, and vegetables," making it impossible to separate commodity types and individual commodities into the other two produce categories (vegetables and fruits and nuts). Some growers reported production of crops and animal products that fell into more than one principal commodity group category.

From the farm address given in the registration form, the county location of each entity was identified. The state's counties were divided into eight geographic regions based on similar groupings used by CDFA in reporting (Fig. 1). The Central Coast and Bay area were combined to maintain grower confidentiality in the Bay Area. Gross sales include only sales of commodities marketed as organic and not sales of commodities that were produced using organic practices but sold into the conventional market. Acres are producing acres and not cropped acres or farm acres. In other words, farmers report land that is double cropped as a single production acre and do not include land that is used as a roadway, fallow, or for buildings.

There were seven third-party certifiers active in California in 1997. Three of these each certified 1% or less of the certified organic growers. California Certified Organic Farmers (CCOF) was the most important certifier in all years, representing 73% of the certified organic growers in 1997. CCOF is unique in that it is a nonprofit grower-run organization dedicated to providing information about organic practices to its growers and serving as a source for buyers to locate organic products. Because of the unique nature of CCOF's grower relationships and services, a dummy variable was created for CCOF certified (DCCOF) and a second dummy variable for certification by any of the other six certification organizations (DOTHER). The benchmark case is no certification such that both dummy variables take on the value of zero.[3]

IV. A BEHAVIORAL MODEL OF EXIT DECISIONS

Each year, an existing organic farming operation must choose whether to remain organic. A farmer choosing to leave the organic sector may remain in farming or exit farming altogether. We do not observe this choice. What we do observe is the choice about whether to continue farming organically, which is a binary one. As such, we use a Random Utility Model (RUM) with two discrete alternatives to model organic farming exit decisions (McFadden, 1974). The motivation for a RUM in this setting is threefold. First, we do not observe profits directly, nor do we observe the mapping of farmer characteristics into profits. However, we expect the RUM to capture the relationship between these characteristics and farmer choice. Second, we expect that some

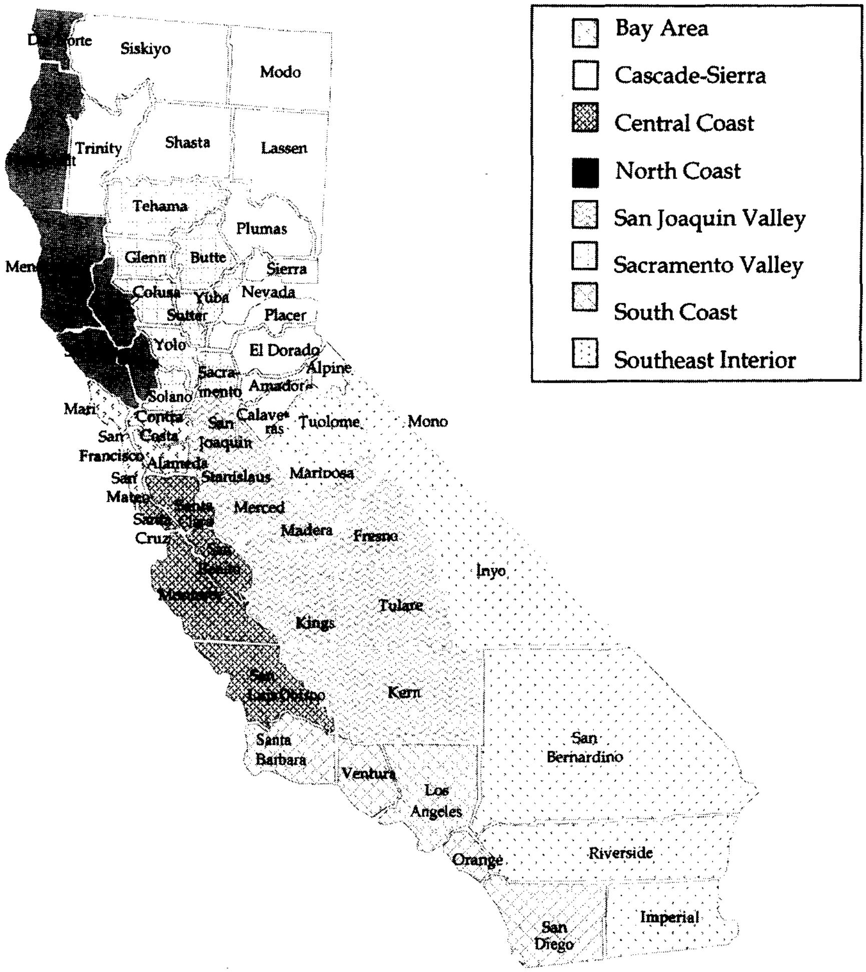

Fig. 1. Regions of Organic Production in California.

characteristics that affect profits, and hence influence choice, are known to the individual farmers but unobservable to the analyst. The RUM presumes that these characteristics are distributed randomly across farmers and choice occasions. Finally, different farmers may actually derive different utilities from farming organically. This, in part, explains behavioral heterogeneity across farmers with similar observable characteristics.

Suppose that a farmer's utility from growing organic is composed of a linear and additive function of characteristics that are observable to the analyst and characteristics that are unobservable to the analyst. Thus, the utility of remaining in organic farming is:

$$\begin{aligned} U_{itm} &= v_{it} + \varepsilon_{im} \\ &= \mathbf{X}_{it}\boldsymbol{\beta} + \varepsilon_{im}, \end{aligned} \tag{1}$$

where i indexes the farmer, t indexes time, m is an indicator variable ($m=1$ for stay in the organic market and $m=0$ for exit), $\mathbf{X}$ contains observable characteristics that differ across individual and/or time, $\boldsymbol{\beta}$ is a parameter vector, and ε contains unobservables. Since we do not observe exiters *ex post*, we assume that the utility of exiting is a constant plus unobservables, that is for $m=0$:

$$U_{itm} = \alpha + \varepsilon_{itm}. \tag{2}$$

From Eqs (1) and (2), we can write the probability of remaining in organic farming, p_{it1}:

$$\begin{aligned} p_{it1} &= Pr(U_{it1} > U_{it0}) \\ &= Pr((\mathbf{X}_{it}\boldsymbol{\beta} - \alpha) > (\varepsilon_{it0} - \varepsilon_{it1})). \end{aligned} \tag{3}$$

By assuming further that ε is distributed iid Type I Extreme Value, we have the familiar logit model for binary choice. In essence, this means that we assume unobservable characteristics that affect the utility of staying or exiting are distributed randomly across individuals, time, and the two choices. As such, Eq. (3) can be written as:

$$p_{it1} = \frac{e^{\mathbf{X}_{it}\boldsymbol{\beta} - \alpha}}{1 + e^{\mathbf{X}_{it}\boldsymbol{\beta} - \alpha}} \tag{3a}$$

An individual makes a sequence of choices and does not necessarily remain in the data set for the same amount of time as another individual. Once a farmer exits, we assume that there are no more choice occasions.[4] Moreover, new entrants may have fewer choice occasions. We can accommodate this phenomenon by considering the likelihood of observing each individual's sequence of choices. Figure 2 depicts possible sequences of farmer decisions. Let y_{it} be an indicator variable for staying in organic farming, let $t_{0i} = \max\{i\text{'s}$ entry year, 1993$\}$, and let $\tau_i = \min\{i\text{'s}$ exit year, 1997$\}$ because we only have data on exit decisions from 1993–1997. The likelihood for farmer i's sequence is:

$$L_i = \prod_{t=t_{0i}}^{\tau_i} (p_{it1})^{y_{it}} (1 - p_{it1})^{1-y_{it}}. \tag{4}$$

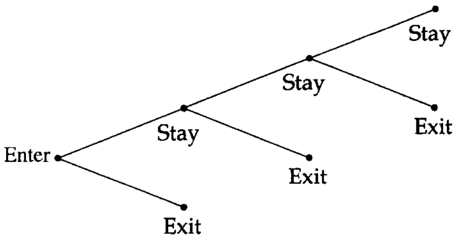

Fig. 2. Farmer Decision Sequence.

The likelihood for the entire sample is then:

$$L = \prod_{i=1}^{n} \prod_{t=t_{0i}}^{\tau_i} (p_{it1})^{y_{it}} (1 - p_{it1})^{1-y_{it}}. \tag{5}$$

Thus, the model can be estimated using maximum likelihood simply by summing the log-likelihoods of individual observations.

The choice between remaining in organic farming or exiting, either by switching to conventional farming or giving up farming altogether, involves the relative profitability of all options tempered by individual preferences. At the very least, a decision-maker compares expected net returns from farming organically to expected net returns from conventional farming. A variety of market and farmer-specific characteristics, in turn, influence these expected net returns. Looking first at income, price premia for organic products vary substantially across crops and time. Further, these premia can only be obtained though access to certain organic markets. There is considerable uncertainty about when and where these price premia materialize.[5] Individual farm soil characteristics, availability of water resources, climate, and weather influence the relative yields from organic and conventional growing methods. Moreover, these influences are heterogeneous across commodities and locations. Of course, individual farmer skill and knowledge of different growing practices affect yields as well. Turning to the cost considerations, the relative costs of conventional and organic farming are determined by the input mix for each of the systems and the corresponding prices of those inputs. Dissimilarity between conventional and organic input mixes is heterogeneous across commodity and geographical space, ultimately leading to cost differences between conventional and organic that vary by commodity and location.

Although individual farmer preferences about organic agriculture and expectations about net returns are unobserved in the organic registration data, and arguably unobservable, the observed characteristics in the data set are indications of relative profitability and preferences for organic farming. The observable characteristics in **X** are farm size (ACRES and SALES), organic certification (DCCOF and DOTHER), regional dummies (DR1–DR7), commodity group dummies (DC1–DC6), and years in the data set (YR_IN).[6]

ACRES and SALES are both measures of farm size, which, in turn, partially determine the net returns for the farm. Larger farms may have more access to certain markets, particularly the rapidly growing organic processed foods market. Farm size may also enhance an operator's ability to obtain premia for organic foods consistently, improving an operator's position to bargain with processors, wholesalers, and retailers with market power. Also, many buyers want to obtain a minimum quantity from an individual grower, effectively locking out smaller growers who cannot meet these requirements. Larger organic farms could also experience some returns to scale.

DCCOF and DOTHER may indicate access to information and markets as well as farmer preference for the practice of farming organically. Certifying agencies may provide information in various formats such as newsletters, in-house publications, educational meetings, or field days. Many wholesalers, processors and retailers require certification as a condition of purchasing organic products. This requirement creates paper trails for product sources and consequently serves as a legal safeguard. Lohr and Salomonsson (2000) find that access to information and markets can substitute for direct payments to farmers in Sweden and encourage organic farming. Thus, our certification dummies may reflect the value of information and access to markets, which influence the relative returns from organic agriculture. At the same time, paying certification organizations is also a way of supporting the grassroots organic farming movement, and hence may reflect a willingness to forego some financial opportunities in order to maintain an ethical commitment to sustainable agriculture.

Farm location and crop choice also affect the relative profitability of conventional and organic farming. In this chapter, regional dummies represent the following:

DR1	North Coast
DR2	Central Coast + Bay Area
DR3	Sacramento Valley
DR4	San Joaquin Valley
DR5	South Coast
DR6	South Eastern Interior
DR7	Cascade-Sierra

Ex ante, we expect that access to markets for organic products differs widely across these regions, and thus the expected net returns from conventional and organic farming differ. Specifically, we expect proximity to the large urban areas of San Francisco, Los Angeles, and San Diego to show the fastest rates of growth, lowest exit rate, and highest rates of entry. The majority of organic handlers are located in either the San Francisco Bay area or the Los Angeles area.

Commodity group dummies represent the following:

DC1	Field Crops
DC2	Fruits and Nuts
DC3	Livestock
DC4	Nursery
DC5	Vegetables
DC6	Unclassified produce

We expect that both expected price and yield differentials between conventional and organic to vary by crop. This collection of dummy variables provides some proxy for these differences.

The YR_IN variable may signal a long-term plan for organic production, an accumulation of experience and information about organic production, and a moral commitment to organic farming. As such, this variable captures aspects of profitability and individual farmer preference.

Table 3 reports the results of two binary logit regressions. There are 6,540 observations, of which there are 1,490 exit decisions. The χ^2 statistics reported are for restricting all of the slope coefficients to zero. In all cases, we strongly reject this hypothesis. The first model contains regional dummies and commodity group dummies. Note that DC2 and DR7 were dropped for purposes of avoiding perfect multicollinearity in the dummy variables. As such, the model's benchmark is for fruit and nut growers in the Central Coast or Bay Area. The second model imposes restrictions based on the test results of the first model.

In the first model, both indicators of farm size are positive but insignificant. The lack of significance is likely due to collinearity between SALES and ACRES. Both certification dummies are positive and strongly significant. As such, the probability of remaining in the organic sector is higher when the farmer is certified by an outside certification organization. The coefficients are also strikingly similar.

None of the regional dummies are statistically significant, but three commodity group dummies, livestock (DC3), vegetable (DC5) and unclassified produce (DC6) are statistically significant. The livestock dummy is negative and significant. This suggests that organic livestock producers are more likely to exit than the benchmark producer (fruits and nuts). Livestock operations make entry and exit decisions that are tied to the life cycle of the animals and not to the land. Most of the production under the livestock category is poultry and eggs which does not involve pasture. Thus, the result is sensible because organic capital no longer exists once animals are sold.

The vegetable coefficient is also negative, though smaller in magnitude. This result is at first glance curious because organic capital is fungible when vegetables are harvested; growers can substitute among different organic vegetable crops each year. In the short run, fruit and nut growers are locked into perennial crops and across years can only substitute between conventional and organic versions of the crops. However, moving out of organic production and

Table 3. Behavioral Model of Organic Farming Exit Decisions.

Dependent Variable = 1 when the farmer remains organic

	Coefficient	s.e.	Z-Stat.	Coefficient	s.e.	Z-Stat.
INTERCPT	0.3683	0.135	2.734***	0.4736	0.065	7.286***
ACRES	0.000924	0.001	1.621	0.00173	0.001	3.399***
SALES	1.86E-07	1.95E-07	0.956			
DCCOF	0.7213	0.079	9.189***	0.6611	0.073	9.044***
DOTHER	0.7918	0.154	5.158***			
DR1	0.0763	0.133	0.575			
DR2	0.091	0.141	0.647			
DR3	0.2031	0.155	1.310			
DR4	0.0559	0.172	0.326			
DR5	0.0792	0.131	0.606			
DR6	–0.0633	0.171	–0.371			
DC1	–0.1958	0.194	–1.009	–0.0919	0.185	–0.496
DC3	–1.0134	0.332	–3.056***	–1.0178	0.331	–3.076***
DC4	0.1987	0.268	0.741	0.1934	0.266	0.726
DC5	–0.1535	0.075	–2.039**	–0.1757	0.068	–2.572**
DC6	0.2901	0.115	2.518**	0.2598	0.112	2.328**
YR_IN	0.2359	0.025	9.474***	0.2491	0.025	10.044***
	$\chi^2(16)$	345.41***		$\chi^2(8)$	308.605***	

Notes:

*** indicates one-side t-statistic is significant at the 1% level,** at the 5% level, and* at the 10% level.

DCCOF and DOTHER are dummy variables for organic certification.

DR1–DR6 are regional dummy variables, and DC1–DC6 are commodity group dummy variables.

into conventional farming means that there will be at least a three year transition period before the grower can resume organic sales. Therefore, substituting conventional vegetable production for organic vegetable production results in a waiting period comparable to the time period needed to bring a newly planted organic orchard into production or transition a conventional orchard to organic. It may be that the markets for organic vegetables are more volatile and easily saturated than those for organic fruits and nuts, explaining the higher propensity of vegetable growers to exit than fruit and nut growers. The unclassified produce is positive and significant. As such, highly diversified organic produce farms are more likely to persist.

Finally, the YR_IN coefficient is positive and strongly significant. Put simply, the more years that the farm is organic, the more likely it will remain organic. This is consistent with the hypothesis that organic operators build long run reputations as part of their marketing strategies and are likely to be able to maintain their markets once they have developed them. This result implies that it might be difficult to break into the organic market.

Three additional joint hypotheses are tested with this first model. First, we test to see if the certification variables are statistically different. The test statistic, distributed $\chi^2(1)$, is 0.1962. Thus, we fail to reject this restriction and conclude that there is no difference in the effect of different certifying organizations on the propensity to exit. The important distinction is simply whether or not a farmer is certified by one of the organizations. Second, we test that the regional dummies are jointly equal to zero. The test statistic, distributed $\chi^2(6)$, is 3.3211, so we fail to reject again. This is a somewhat surprising finding because it suggests that there are no unobserved regional characteristics that influence organic farming exit decisions. It also may be an indication that markets for organic produce are spatially integrated. That is, there are no organic marketing advantages based on a farm's geographical locations. Nonetheless, we would expect the regional dummies might reflect regional differences in climate and pest pressure. However, these differences are more likely to be at a subregional level and not captured by our broad regional scheme. Third, we test whether the commodity dummies are jointly equal to zero. The test statistic, distributed $\chi^2(5)$, is 25.1722, so here we actually reject this set of restrictions.

Based on the hypothesis tests above and a desire to resolve the impact of farm size on exit probability, a second model is estimated. This model drops SALES, DOTHER, and the regional dummies. Qualitatively, the results are very similar to those of the first model. All of the signs are the same, and coefficient magnitudes are similar. Here, the ACRES coefficient is strongly significant, indicating a propensity for smaller organic farms to exit.[7]

V. CHARACTERISTICS OF ENTRANTS

Since the growth in California's organic sector is composed of many entrants each year offset by somewhat fewer exiters, it is important to analyze the characteristics of entrants to assess the future of organic farming in the state. Do the characteristics of entrants into the organic sector differ from those of incumbents, and if so, how do they differ? To answer these questions, we use several tests for differences in means between incumbents and entrants. First we compute t-tests for differences in means of continuous random variables, including total acreage and total sales. Next we compute similar t-tests for differences in the binary certification variable. Finally, we present a series of binary choice models to assess differences in regional and commodity group characteristics.

Difference in means tests strongly indicate that incumbent organic farms are larger than entrants in every year. Table 4 reports t-tests for differences in mean acreage and mean sales between incumbent and entrant farms on an annual basis. These t-statistics are computed according to the following:

Table 4. Differences in Means – Continuous Variables.

	1993	1994	1995	1996	1997
Incumbent					
n	796	1024	1124	1134	1193
Mean Acres	42.03	38.36	35.08	34.93	41.58
s.e. Acres	129.17	124.22	113.92	123.77	179.02
Mean Sales	84,442	83,313	84,916	102,185	109,856
s.e. Sales	390,488	396,727	389,041	488,504	496,156
Entrant					
n	333	348	300	335	335
Mean Acres	16.20	13.85	17.55	30.88	29.66
s.e. Acres	59.76	52.93	62.02	145.97	85.32
Mean Sales	23,654	17,453	46,977	38,985	24,922
s.e. Sales	116,937	83,727	347,748	278,189	65,913
t-Acres	4.59***	5.10***	3.55***	0.46	1.71**
t-Sales	3.99***	4.99***	1.64*	3.02***	5.75***

Note:
*** indicates one-side t-statistic is significant at the 1% level, ** at the 5% level, and * at the 10% level.

$$\frac{\bar{X}_{Incumbent} - \bar{X}_{Entrant}}{\sqrt{\frac{s^2_{Incumbent}}{n_{Incumbent}} + \frac{s^2_{Entrant}}{n_{Entrant}}}}. \tag{6}$$

In all years, total sales are larger and statistically significant for incumbent farms compared to entrant farms. Acreage shows a similar relationship, though it is not statistically significant in one of the five years tested.

Table 5 reports tests for differences between incumbents and entrants in the share of organic farmers that are certified. The formula is the same as Eq. (6), but the variance estimate for the Bernoulli variables is simply $\hat{p}(1-\hat{p})$. A more conservative way of computing this variance is to consider the maximum possible variance being at $\hat{p} = 0.5$. Both sets of t-statistics are reported. In all years, the share of certified organic farmers is greater among incumbents and statistically different from entrants. This is consistent with the results of the exit analysis. Farmers that become certified are less likely to exit organic farming. Many farmers enter without having been certified, but most of the farmers that remain organic become certified eventually.

Analyzing differences in commodity groups and regional variation is somewhat more complicated than simple pair-wise testing of means. Here we

Table 5. Differences in Means – Share of Certified Organic Farmers.

	1993	1994	1995	1996	1997
Incumbent					
n	796	1024	1126	1136	1198
Share Certified	0.544	0.472	0.434	0.436	0.451
Var. Share Certified	0.248	0.249	0.246	0.246	0.248
Var. Maximum	0.250	0.250	0.250	0.250	0.250
Entrant					
n	333	348	301	339	335
Share Certified	0.192	0.098	0.150	0.233	0.278
Var. Share Certified	0.155	0.088	0.127	0.179	0.201
Var. Maximum	0.250	0.250	0.250	0.250	0.250
t-Share(calculated s.e.)	12.61***	16.78***	11.25***	7.43***	6.10***
t-Share(maximum s.e.)	10.78***	12.05***	8.78***	6.55***	5.60***

Note:
*** indicates one-side t-statistic is significant at the 1% level, ** at the 5% level, and * at the 10% level.

use a series of binary choice models to test for differences between incumbents and entrants. It is important to note that though the statistical models are essentially the same as those in the exit decision model, they are not behavioral models. They are a convenient way to formulate joint hypotheses about certification, regional, and commodity qualitative variables. In all cases, the dependent variable captures incumbent or entrant. Thus, positive (negative) coefficients indicate that the characteristic is higher (lower) for incumbents.

Table 6 reports results for differences in type of certification. DCCOF indicates that a farmer has been certified by California Certified Organic

Table 6. Differences in Qualitative Variables for Entrants and Incumbents Certification Organizations.

	Coefficient	s.e.	Z-stat.
1993			
Intercept	0.30	0.08	3.70***
DCCOF	1.80	0.18	10.28***
DOTHER	0.82	0.28	2.92***
$\chi^2(1)$-Wald		9.70***	
1994			
Intercept	0.54	0.07	7.67***
DCCOF	2.39	0.23	10.39***
DOTHER	1.18	0.32	3.69***
$\chi^2(1)$-Wald		9.88***	
1995			
Intercept	0.91	0.07	12.31***
DCCOF	1.57	0.19	8.10***
DOTHER	1.11	0.33	3.37***
$\chi^2(1)$-Wald		1.54	
1996			
Intercept	0.9065	0.0735	12.333333***
DCCOF	1.0732	0.1626	6.600246***
DOTHER	0.4719	0.2311	2.0419732***
$\chi^2(1)$-Wald		5.1931**	

Note:
*** indicates one-side *t*-statistic is significant at the 1% level, ** at the 5% level, and * at the 10% level.

Farmers. DOTHER indicates that a farmer has been certified by another organization, e.g. QAI or FVO. Some farmers have multiple certifications, which coincides with both DCCOF = 1 and DOTHER = 1. For all regressions, the coefficients on both dummy variables are positive and statistically significant. Thus, the share of incumbents certified by either or both type of organization is higher than the share of entrants. In all years, the coefficient on DCCOF is larger than that on DOTHER, suggesting that the share of incumbents that are CCOF is greater than the share that are certified by another organization. This difference is statistically significant in four of the five years reported, as indicated by the χ^2 tests.

Table 7 reports similar analyses for regional dummies. The coefficient signs indicate the way that regional patterns differ between incumbents and entrants. To avoid a dummy variable trap, DR7 was dropped. Thus, region 7 is the benchmark for the intercept term. Only some of the individual coefficients are statistically significant, and signs change across years. For instance, DR5 is negative and significant in 1995 but positive and significant in 1996. More importantly, the χ^2 tests (with six degrees of freedom) show that the regional pattern of entrants is not the same as the regional pattern of incumbents in every year. With the exception of 1996, entrants are more likely to go into the South Coast than into other regions. Incumbents are more likely to be in the Sacramento Valley, though the coefficient is only significant in one year.

Table 8 reports similar analyses for commodity group dummies. In this case, we use commodity group 2, fruits and nuts, as a benchmark. The coefficients on DC6 are positive in all years and significant in three of the four years, suggesting that incumbents are more likely to have a mixture of fruits, nuts, and vegetables than entrants. This, in turn, suggests that incumbents diversify their portfolios of organic crops after entering with fewer types of crops. In the last three years, DC5 is negative and significant in two of these years. This indicates that entrants are relatively more likely to grow vegetables. Curiously, the sign is reversed in 1993. As in the previous analyses, the χ^2 tests indicate that there are statistical differences in the share pattern between incumbents and entrants. Here the tests have five degrees of freedom and are significant in three of the four years.

VI. DISCUSSION

Although there seems to be consensus that the organic industry will continue to grow, predictions about the percentage of market share that organic will capture in the years following implementation of the USDA labeling rules range from two to 10%, and there is considerable uncertainty about the future

Table 7. Differences in Qualitative Variables for Entrants and Incumbents Regional Dummies.

	Coefficient	s.e.	Z-stat.
1993			
Intercept	1.04	0.27	3.80***
DR1	−0.04	0.31	−0.14
DR2	0.41	0.33	1.22
DR3	0.27	0.35	0.78
DR4	0.83	0.41	2.01**
DR5	−0.86	0.29	-2.91***
DR6	0.18	0.42	0.43
$\chi^2(6)$-Wald		67.19***	
1994			
Intercept	0.89	0.23	3.91***
DR1	Q.29	0.27	1.09
DR2	0.38	0.29	1.35
DR3	0.92	0.33	2.81***
DR4	0.52	0.33	1.55
DR5	−0.12	0.25	−0.48
DR6	−0.07	0.34	−0.21
$\chi^2(6)$-Wald		24.38***	
1995			
Intercept	1.50	0.27	5.58***
DR1	−0.05	0.31	−0.18
DR2	0.14	0.33	0.41
DR3	0.31	0.36	0.86
DR4	0.28	0.39	0.73
DR5	−0.50	0.29	−1.73*
DR6	−0.55	0.36	−1.52
$\chi^2(6)$-Wald		23.02***	
1996			
Intercept	0.97	0.23	4.13***
DR1	0.11	0.27	0.41
DR2	0.16	0.28	0.56
DR3	0.39	0.31	1.29
DR4	0.46	0.34	1.37
DR5	0.45	0.26	1.73*
DR6	−0.31	0.31	−0.98
$\chi^2(6)$-Wald		14.00**	

Note:
*** indicates one-side *t*-statistic is significant at the 1% level, ** at the 5% level, and * at the 10% level.

Table 8. Differences in Qualitative Variables for Entrants and Incumbents Commodity Group Dummies.

	Coefficient	s.e.	Z-stat.
1993			
Intercept	0.71	0.08	8.68***
DC1	1.12	0.47	2.38**
DC4	0.15	0.60	0.25
DC5	0.29	0.16	1.78*
DC6	0.55	0.22	2.55
Note that DC3 was dropped because there were no organic livestock entrants in 1993.			
$\chi^2(5)$-Wald		13.47**	
1994			
Intercept	1.11	0.08	13.70***
DC1	0.69	0.42	1.64*
DC3	–0.05	0.83	–0.06
DC4	–0.10	0.59	–0.16
DC5	–0.25	0.14	–1.75*
DC6	0.15	0.20	0.76
$\chi^2(5)$-Wald		6.89	
1995			
Intercept	1.31	0.08	15.70***
DC1	0.42	0.42	1.01
DC3	–0.23	0.75	–0.30
DC4	–0.16	0.48	–0.34
DC5	–0.23	0.14	-1.57
DC6	0.74	0.27	2.73***
$\chi^2(5)$-Wald		12.78**	
1996			
Intercept	1.41	0.08	16.61***
DC1	–1.08	0.26	–4.10***
DC3	–2.24	0.61	–3.67***
DC4	–0.05	0.39	–0.12
DC5	–0.46	0.14	–3.31**
DC6	0.47	0.28	1.67*
$\chi^2(5)$-Wald		43.53***	

Note:
*** indicates one-side *t*-statistic is significant at the 1% level, ** at the 5% level, and * at the 10% level.

composition of organic agriculture. Our analysis demonstrates that the characteristics of entrants and incumbents are quite different. This work also clearly demonstrates that organic farm characteristics are important determinants of whether an individual farm exits the organic sector. As the market grows, the composition of farmers supplying organic products will continue to evolve with respect to farm size, location, and commodities grown. Changes in the composition of marketing outlets for organic foods, the growth in consumer demand by food category combined with the development of new processed products, and the response to the USDA organic standards will all affect the demographics of organic farming.

The evolution of marketing channels will likely influence the size distribution of organic growers. Expansion of natural food store chains, increased organic sales in conventional grocery stores, and internet sales certainly could expand the total market for organic products but may not benefit all producers equally. The 1997 national survey of organic farmers by the Organic Farming Research Foundation revealed that 13% of respondents' organic commodities were marketed directly to consumers, 80% through wholesale, and 7% directly to retail outlets. The future distribution of sales by market type ultimately will depend on the marketing outlets through which new consumers enter the organic market and veteran consumers expand their purchases. A concern in the organic community is that smaller growers will be unable to take advantage of these future marketing opportunities.

While we do not observe them on the organic registration forms, marketing outlets are typically different for different size growers. Small growers mostly rely on direct sales, including farmers markets, roadside stands, and Community Supported Agriculture (CSAs).[8] Larger growers mainly sell through wholesalers and directly to retailers.[9] As in conventional agriculture, growers must be able to supply a minimum amount of product consistently to sell in these markets. Therefore, the composition of the marketing outlets as well as overall growth are essential to the staying power and entry of farmers with different scale operations. In our analysis, the tendency for higher attrition among small farmers may actually reflect changes in marketing channels that are already underway.

Our study suggests that entrants are more likely to be smaller than the population of incumbent organic growers but at the same time smaller growers are more likely to exit the organic market. If we assume that smaller growers are more likely to direct market to consumers, then they are the group most vulnerable to changes in market share among market outlets, since they do not have access to mass market outlets and, to a lesser extent, natural food store outlets. Our results do not find a higher staying power for growers closer to large population areas in California nor do we see higher rates of entry in certain areas of California over others. This suggests that direct market channels are well developed throughout the state and that growers are able to find market outlets in all regions of California. Thus, smaller growers may survive in the face of a shrinking market share for direct marketing if the absolute size of direct marketing outlets at least remain constant and larger growers do not displace smaller growers in these venues.

Although growth in the organic market has varied across food categories, fresh produce remains the largest category of organic food, capturing 40% of organic food sales nationally. Produce is more likely to be direct marketed to consumers than other categories of organic foods. Within the produce category, 11% is direct marketed to consumers compared to 6% for all organic foods, according to a study by the Nutrition Business Journal (2001). Nonetheless, 47% of produce is sold through supermarkets and 42% through natural food stores. Entrants into the organic market are more likely to grow vegetables than any other commodity type, while vegetable growers are also most likely to exit. This finding undoubtedly reflects the intense competition in the marketing of organic vegetables in all venues.

Given a highly competitive organic vegetable market, entry of a single large organic grower has the potential to displace several small growers. Further, a large conventional grower with both organic and conventional products might be in a better position to market organic product into a conventional market, as these relationships are already established. Market saturation is a concern that is often expressed by those within the organic industry at all levels of production. Anecdotal evidence suggests that some growers disc in commodities or sell in the conventional market when they are unable to find a home for their products in the organic market. Eventually market saturation may lead to exit from the organic market and possibly from farming altogether for displaced growers.

The product mix of processed organic foods will also have an impact on the future composition of organic farming. New items continue to appear within entrenched organic categories such as milk, soymilk, frozen vegetables, and frozen entrees. The multiplicity of organic canned foods entering the marketplace reflects consumer demand for convenient healthy meals. Expanded lines of packaged organic cereals as well as snack and candy items using organic inputs also indicate consumer preferences for non-GMO grains. New processed food items change the demand for both the quantity and diversity of organic commodities. As with access to grocery stores and large natural food stores, processors seek business relationships with growers who can supply the required quantity on a consistent basis.

Arguably the provisions in the OFPA that will lead to the greatest change in the organic industry are those related to certification. Growers grossing over \$5,000 will be required to be certified in order to market using an organic label. In California about 400 growers who were not certified in 1997 would require certification under the new federal rule. In any given year, over 300 new growers register with the state of California Organic Program. The results of our analysis show that entering growers are likely to have lower incomes than incumbent growers. When the federal rules requiring certification are implemented, there will be hundreds of low income growers seeking certification for the first time. The challenge to the certification organizations will be to serve this new clientele in a manner that is economically viable for the grower and the certifier without compromising the integrity of organic certification. The long term role of federal and state governments in cost share programs for certification has yet to be determined. USDA provided funds to 15 states in 2001 to share the cost of organic registration for growers already certified in 2000. Several states such as Nevada and Kentucky provide certification services at below the cost of service.

Perhaps more importantly is whether the OFPA guidelines exacerbate disadvantages faced by small organic growers. As in the status quo, there is no program for growers grossing under \$5,000. These growers may be at a disadvantage in the marketplace when their products are marketed side-by-side with growers of certified organic products. The consumer may perceive a higher quality associated with the certified product. Unlike the status quo, however, all growers with over \$5,000 in sales will have to be certified by a USDA accredited certifier under the new rule. New certification organizations undoubtedly will emerge, some existing certifiers will expand their geographic purview, and some certifiers will cease to operate. Of concern is whether or not the cost of certification will be prohibitive to low income growers above the \$5,000 sales figure. At the same time, many growers who are exempt from the certification requirement but who have been certified in the past fear that certifiers will no longer want to certify them because of the relatively high cost of certifying a small grower.

Another important aspect of the new federal regulations is that certifiers are prohibited from supplying individualized information to clients to help them achieve certification because this practice is viewed as consulting and therefore a conflict of interest. The certifiers are not prohibited from educational activities available to the general public. However, the provision of tailored information by many certifiers has been an important service provided to clients and possibly the reason for selecting one certifier over another. The analysis here concludes that growers are more likely to stay in organic production if they are certified. If growers stay in organic production at least in part due to the production information provided by their certifier, then this new restriction on the services provided by certifiers could work to increase the attrition rate of organic growers. Again, it is worth asking whether this rule will favor large growers that presumably are less dependent on information from their certifiers.

Organic agriculture in California grew rapidly in the 1990s. This growth was characterized by many new entrants and many exiting farms in each year. As a result, the composition of organic farming evolved substantially. Although the future size and character of the organic industry in California and throughout the U.S. are unknown, growth and continued change seem inevitable. This transformation has salient implications for issues that transcend organic agriculture, including farmland preservation, sustaining family farms, economic viability of rural communities, food safety, and water quality.

NOTES

1. Livestock, poultry, and products includes dairy products and eggs.
2. California is not one of the states included in the cost-share program. The states included are Connecticut, Delaware, Maine, Maryland, Massachusetts, Nevada, New Hampshire, New Jersey, New York, Pennsylvania, Rhode Island, Utah, Vermont, West Virginia, and Wyoming.
3. Because some growers are certified by multiple organizations, it is possible that both DCCOF and DOTHER take on the value of one.
4. In fact, it is possible for a farmer to exit the program and then re-enter at another point in time either on the same farm or by purchasing or leasing a farm that has already gone through the organic transition period required. However, the data did not reveal any such cases and we did not include this possibility in the model.
5. In a survey of Canadian organic farmers, Henning, Baker and Thomassin (1991) find that many farmers express concerns about the unreliability of organic premia, even though the farmers report 30% premia on average.
6. Using a similar empirical model, Kimhi and Bollman (1999) study family farm exits in Israel and Canada. Though these authors deal with the choice between farming and not farming and do not capture the possibility of switching to another farming activity, their analysis contains many of the same variables that we use here, including commodity and regional dummies and farm size. Interestingly, they find that farm size has a positive effect on exit probability for Israel but a negative one for Canada.
7. We re-ran this model including SALES and dropping ACRES and found qualitatively similar results. In particular, the SALES coefficient was negative and significant. In general, the two variables are collinear, suggesting why neither was individually statistically significant in the first model. The second model reports ACRES rather than SALES because ACRES provides a better indicator of the size of an organic operation given that organic certification is tied to the land and not to the grower.

8. Community Supported Agriculture is equivalent to subscription agriculture where the consumer pays a monthly fee and receive a bundle of seasonal produce from a local grower typically on a weekly basis.

9. In fact, the difference in marketing outlets was part of the rationale for exempting growers grossing less than $5,000 from obligatory certification.

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8. ORGANIC FARMING POLICY IN THE EUROPEAN UNION

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ABSTRACT

Organic farming is recognised in the European Union as one possible model to improve the sustainability of agriculture. During the 1990s the sector grew rapidly (to 3% of agricultural area in 2000), caused in part by policy support measures in member states and the EU. The paper summarises the development of the organic sector, discusses reasons for policy support, and reviews the main policy measures at EU and country level in three areas: legislation defining organic production, direct payments and other measures. It concludes that in future the integration of policy measures within countries and at EU level should be improved, in particular through the development of national and European action plans for organic farming.

1. INTRODUCTION

Organic farming aims to create an integrated, humane, environmentally and economically sustainable agricultural production system. In order to provide acceptable levels of crop, livestock and human nutrition, protection from pests and diseases, and an appropriate return to the human and other resources employed, maximum reliance is placed on self-regulating agro-ecosystems, locally or farm-derived renewable resources, and the management of ecological

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and biological processes and interactions. Reliance on external inputs, whether chemical or organic, is reduced as far as possible. The term 'organic' is best thought of as referring to the concept of the farm as an organism, rather than the type of inputs used.

In practice, and for the purposes of this study, organic farming is defined by European Union (EU) legislation, in particular Regulations 2092/91 and 1804/1999 (see below). These regulations are substantially equivalent to recent USDA regulations (USDA, 2000) as well as the internationally recognised Codex Alimentarius (FAO, 1999) and International Federation of Organic Agriculture Movements standards (IFOAM, 2000) and therefore represent the generally accepted definition of organic agriculture used for international trade.

Organic farming is increasingly recognised in the EU, by consumers, farmers, environmentalists and policy-makers alike, as one of a number of possible models for environmental, social and financial sustainability in agriculture. The 1990s witnessed very rapid growth in the sector. In 1985, certified organic production accounted for just 100,000 ha on 6,300 holdings in the EU, or less than 0.1% of the total utilisable agricultural area (UAA). By the end of 2000, this had increased to nearly 4 million hectares (Table 1) on 139,282 holdings, or nearly 3% of total UAA and 2% of holdings (see Foster & Lampkin, 2000, for detailed statistics for 1993–1998). The figures hide great variability within and between countries. By 1999 several European countries had achieved 3–9% of their agricultural area managed organically, rising to more than 30% in some regions within countries. In Austria (AT), for example, 9% of UAA was managed organically (see Fig. 1). Other countries with relatively high percentages of holdings and land area converted to organic production are Denmark (DK), Finland (FI), Italy (IT) and Sweden (SE). Germany (DE) and the United Kingdom (UK) are similar to the EU average. However, many others are still at or below the 1% level, with the lowest rates of organic farming to be found in Ireland (IE), Luxembourg (LU) and Greece (GR).

Growth trends in individual countries have varied considerably, with periods of rapid expansion followed by periods of consolidation and, occasionally, decline. However, the overall growth rate in the European Union has been consistently around 25% per year for the last ten years, i.e. exponential growth, with no indication of a decline so far. A projection of these growth rates forward to 2010 gives some indication of the potential significance of organic farming within a relatively short period. Assuming a starting point of 2.0% of EU agriculture in 1998, continued 25% growth each year would imply a 10% share by 2005 and nearly 30% by 2010. It is unlikely that the 25% growth rate

Table 1. Development of Certified and Policy-Supported Organic Land Area in Europe (1985–2000) in ha × 10E3.

Year ending	*85*	*86*	*87*	*88*	*89*	*90*	*91*	*92*	*93*	*94*	*95*	*96*	*97*	*98*	*99*	*00*
Austria	6	7	8	12	17	22	28	84	136	192	336	309	345	288	290	267
Belgium	1	1	1	1	1	1	1	2	2	3	3	4	7	12	19	20
Denmark	5	5	5	6	10	12	18	19	20	21	41	46	64	99	147	165
Finland	1	1	1	2	2	7	13	16	20	26	45	85	102	126	137	147
France	45	50	55	60	65	72	81	85	88	95	118	137	165	219	316	370
Germany	25	27	33	42	54	105	188	299	373	445	462	476	450	417	452	546
Greece	0	0	0	0	0	0	0	0	1	1	2	5	10	15	18	25
Ireland	1	1	1	2	4	4	4	5	5	5	13	20	24	29	32	32
Italy	5	6	6	9	11	13	17	30	88	154	204	334	641	786	959	1040
Luxembourg	0	0	0	0	1	1	1	1	1	1	1	1	1	1	1	1
Netherlands	2	3	3	5	7	7	9	10	11	11	13	14	17	19	22	28
Portugal	0	0	0	0	1	1	2	2	3	7	11	9	12	25	48	50
Spain	2	3	3	3	3	4	4	8	12	17	24	104	152	269	352	381
Sweden	2	3	5	9	29	33	38	40	45	55	87	162	205	244	307	372
United Kingdom	6	7	9	11	19	31	34	35	31	32	48	50	106	275	391	527
European Union	100	113	131	162	222	312	439	636	836	1066	1408	1757	2302	2823	3489	3972

Source: Lampkin (2001) and Foster and Lampkin (2000).

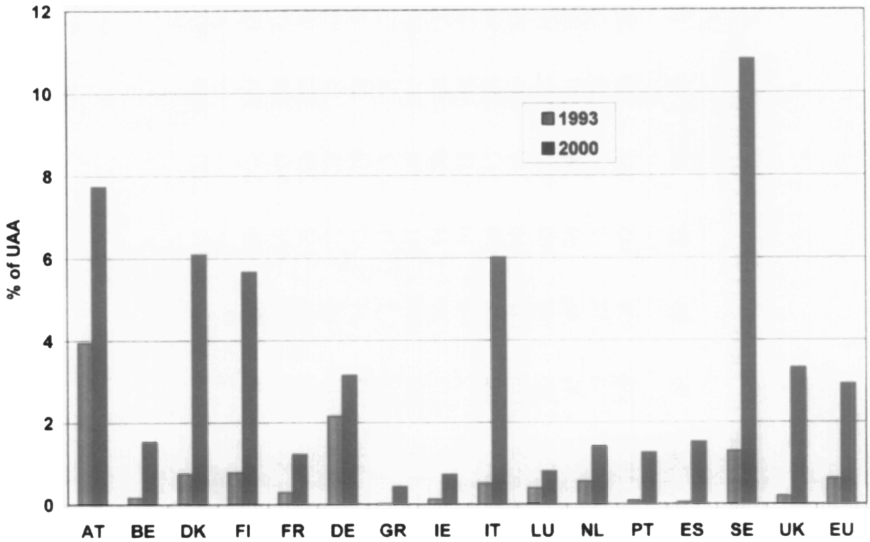

Fig. 1. Development of Organic Land Area (Certified and Policy Supported) as a Percentage of Conventional UAA Between 1993 and 1999 by Country.

will be maintained, but a slower rate of growth of 15% each year would still result in a sector size of just under 5% of agricultural area by 2005 and 10% by 2010.

Alongside the increase in the supply base, the market for organic produce has also grown, but statistics on the overall size of the market for organic produce in Europe are still very limited. The retail sales value of the European market for organic food was estimated to be in the order of EUR[1] 5–7 billion in 1998 (Datamonitor, 1999; ITC, 1999).

A number of reasons can be identified for this recent rapid development, among others the support that organic farming has received through the Common Agricultural Policy (CAP) of the EU. Nearly 80% of the expansion in the land area has taken place since the implementation in 1993 of EC Regulation 2092/91 defining organic crop production and the widespread application of policies to support conversion to and continued organic farming as part of the EU agri-environment programme (EC Reg. 2078/92).

The former, despite many shortcomings (the most important one being the eight year delay until common minimum standards for organic livestock production could be agreed), has provided a secure basis for the agri-food sector to respond to the rapidly increasing demand for organic food across

Europe. The latter has provided the financial basis to overcome perceived and real financial barriers to conversion on the part of farmers. A range of other measures under the EU's Structural Funds, along with some national initiatives, has also been introduced. In 1996, overall spending on organic farming support policies in the EU was estimated to be in excess of 300 M ECU[2] (Lampkin et al., 1999).

This increasing EU policy support during the 1990s has arisen because of a gradual convergence of policy goals with the underlying objectives of organic farming, including environmental protection, animal welfare, resource use sustainability, food quality and safety, financial viability and social justice. Policy-makers also perceive organic farming to contribute to reducing problems of over-production, due to reduced crop yields, stocking rates and area of specific crops produced. Compared with other, more specifically targeted policy measures, organic farming offers three potential advantages: it addresses all (or most of) the policy goals simultaneously, it utilises the market mechanism to support these goals, and it is recognised globally.

The recent reform package of the CAP under Agenda 2000 provides further opportunities for the development of organic farming support, particularly the potential to support organic initiatives under the new rural development regulation (Fischler, 1999). The debate triggered by the discovery of bovine spongiform encephalopathy (BSE) in German cattle in 2000 has also lead to calls for a re-orientation of agricultural policy in Europe towards organic farming (Fischler, 2001; Kuenast, 2001).

The aim of this paper is firstly to explore the reasons for policy support of organic farming in the EU, secondly to review the main policy measures that were implemented at EU level and in individual countries during the 1990s in the key areas of: (a) standards and legislation defining organic production; (b) direct payments to producers; and (c) other measures, and thirdly to highlight some difficulties that were experienced. The paper concludes with examples of the integration of support policies into national action plans. The authors argue that in future more integrated policy development at EU level in line with the new Rural Development Regulation of Agenda 2000 could address some of the shortcomings identified.

The paper is based on work carried out as part of a research project on organic farming and the Common Agricultural Policy (CAP). The overall objective of the project was to provide an assessment of the impact of the 1992 CAP Reform and possible future policy developments on organic farming, as well as the contribution that organic farming could make to EU agricultural and environmental policy goals. The results have been published in a series of reports.[3]

2. REASONS FOR POLICY SUPPORT

Since the mid-1980s European policy makers have increasingly developed an interest in organic farming. Reasons for this include:

- the potential environmental benefits, including soil and habitat conservation, increased bio-diversity, reduced soil and groundwater pollution, improved animal welfare, and utilisation of local and renewable resources;
- the existence of a distinct market for organically produced food (and fibre) as a means by which producers can be compensated for internalising external costs that would otherwise be carried by society;
- a contribution to limiting surplus production and reducing the costs for market support mechanisms due to the lower intensity of organic farming;
- increasing consumer demand for organic products, necessitating an increase in the supply base which offers opportunities for income generation and diversification of farms, potential employment generation due to higher labour demands, and a contribution to rural development objectives.

To aid the policy evaluation, the validity of these claims was assessed on the basis of existing research and other material, but in several cases the lack of statistical and other data on the organic sector limited the analysis.

2.1. Environmental Impacts

With regards to potential environmental benefits, a Europe-wide review of published and unpublished research on the environmental impacts of organic farming concluded that "when evaluated on a per hectare scale, organic farming ... has less detrimental effects on the environment and resource use than conventional farming systems" (Stolze et al., 2000, p. 87).

The report adapted the OECD set of environmental indicators for the agricultural sector and reviewed the available evidence with respect to ecosystem, soil, ground and surface water, climate and air, farm input and outputs, animal health and welfare and the quality of the food produced. In no category was a poorer performance of the organic system compared to conventional identified and, in several categories, the performance of organic farming was clearly better. In three areas, climate and air, animal health and welfare, and food quality, there appeared to be a need for further research.

The authors of the report discussed critically whether the chosen calculation of environmental impact on a per hectare basis is the more appropriate scale, as opposed to the unit of output produced. In the latter case organic farming with its comparatively lower output per hectare may not perform as well as in the assessment carried out. However, the review highlighted that results of comparisons of the environmental impact of organic and conventional systems may differ substantially according to farm type and location, which makes the generalization of individual studies for policy purposes problematic, and highlights the need for further research and debate on the issue.

2.2. Market Development

Based on a survey of national experts in 18 countries (who summarised national studies and contacted approximately 10–15 key informants with substantial experience in the market for organic food in their country) Michelsen et al. (1999) found clear evidence of the existence of a distinct market for organic produce in Europe, which is strongly connected to the rapid growth in organic production at the farm level.

Some consumers are willing to pay a premium, although their number may be inversely related to the magnitude of premiums charged at retail level. However, imbalances in the market have been observed, particularly in periods following policy intervention on the supply side. At present there appears to be oversupply for organic meat and milk produce in countries where the majority of converting producers are grassland-based livestock producers, whereas in many countries the demand for fresh produce (i.e. fruit and vegetables) is clearly greater than the supply, leading to substantial imports from other European countries and beyond.

The markets, although developing rapidly, still show characteristics of being immature, which poses challenges for their analysis. Consumers' preferences appear to be changing rapidly in response to availability of organic products, but also to outside stimuli, especially food scares. Overall, the available evidence seems to suggest that there is growing consumer demand for organic produce (e.g. MINTEL, 2000; Datamonitor, 1999; ITC, 1999), but it is not possible to make a reliable prediction of the future potential size of the organic market.

2.3. Surplus Reduction

With regards to surplus reduction, there is clear evidence that crop yields in organic systems are lower than in most conventional systems in the EU, but the outcome of any comparison depends not only on the yields achieved in the organic systems, but also on the intensity of the system with which it is compared (Padel & Lampkin, 1994). Across Europe organic cereals yields are typically in the range of 60–70% of the conventional average, whereas for livestock production yields of between 70 and 100% of conventional systems are reported (Offermann & Nieberg, 2000). In addition, due to the need for fertility building, organic farms grow less arable cash crops in their rotations. It appears therefore very likely that organic farming contributes to the reduction of agricultural surpluses in the EU, but the extent depends on the levels of adoption of organic methods for the particular commodities concerned. Zanoli and Gambelli (1999) attempted to estimate the budgetary impact of the organic farming sector in the EU and came to the conclusion that at least half of the budget spent on organic support would be offset by savings on other measures of the CAP which would have had to be spent in the absence of the same level of organic production. However, they advised some caution in interpreting their results as the analysis was hampered by the lack of statistical data on organic farming.

2.4. Farm Incomes

For agricultural policy makers concerned about farm incomes, the prices received by organic producers are also of concern. Although linked to consumer willingness to pay, there is some evidence that farm-gate prices are not directly related to retail prices and considerable variation according to country and marketing channels has been identified (Michelsen et al., 1999; Offermann & Nieberg, 2000). Economic surveys in a number of EU countries report average organic farm-gate prices for arable crops, across all marketing channels, in the range of 10 to 200% over conventional prices. Price premiums for organic livestock products are generally lower, at 10–30% for milk, 20–30% for beef, while pork and poultry may attract higher premiums. Prices realised via direct marketing may be twice as high as via wholesale channels (Offermann & Nieberg, 2000).

As far as income generation is concerned, a range of one-off surveys of the economics of organic farming have been carried out (for details see Padel & Lampkin, 1994; Offermann & Nieberg, 2000), but only a very small number of EU countries has included a sample of organic farms in their agricultural statistics and regular farm business monitoring, with reporting over a longer period of time. In a review of the available material, Offermann and Nieberg (2000) came to the conclusion that organic and comparable conventional samples achieve on average similar profits, but more often than not, the profits of the organic sample were higher than the conventional comparison group. Nearly all observations were in the range of ±20% of the profits of respective conventional samples.

However, substantial variation was observed depending on farm type and country, with relatively better profitability per Family Work Unit (FWU) on organic dairy farms and higher profits per ha and per FWU on arable farms, whereas too few studies were concerned with horticulture, other grazing and intensive livestock to be able to come to any conclusion. Generally 16–24% of the profit was generated by the subsidy contribution, but access to premium price markets was a very important factor determining profitability. It is likely that since the review was carried out the relative performance of some organic systems will have improved considerably due to reductions in conventional prices.

2.5. Employment and Rural Development

With regards to the potential contribution of organic farming to employment generation and rural development the situation is less clear. Apart from the potentially positive effect on farm incomes, the review of economic surveys estimated additional demand for labour in organic systems to be in the range of 10–20%, but again a considerable variation between countries and farm types was identified (Offermann & Nieberg, 2000), and economic surveys have limitations in not fully considering indirect use of labour, such as through contractors and casual labour, as far as the reporting of labour is concerned (Jansen, 1999). Organic marketing initiatives may have a particular contribution to make to rural development and further research funded by the EU is now in progress, co-ordinated by the University of Wales, Aberystwyth.[4]

Overall the analysis suggests that, with regards to several of the claims made, organic farming can make a valuable contribution to the development of further policies, whereas in other areas further research would be needed to evaluate its contribution to policies and to substantiate claims made.

3. POLICY FRAMEWORK FOR ORGANIC FARMING SUPPORT IN THE EUROPEAN UNION DURING THE 1990s

The 1990s were characterised by a range of European Union initiatives in support of organic farming, which followed on from the engagement of individual countries in this area. After a brief overview of the historical development, the following section summarises the results of a review of the policy and regulatory environment for organic farming support in the EU under the three headings of legal definitions and standards, direct payments and other support programmes (see Lampkin et al., 1999, for detailed results).

Denmark was the first EU country to introduce, in 1987, a law on organic farming which included a legal framework giving guarantees to consumers and offering financial support to producers during the conversion period as well as support for the development of the market and an advisory service (Dubgaard & Holst, 1994). Germany made use of the EU's extensification policy (EC Reg. 4115/88) to introduce support for organic conversion in 1989; France and Luxembourg introduced smaller programmes under the same regulation in 1992. Austria, Sweden and Finland had national conversion support programmes prior to their accession to the EU in 1995, some of which also included other measures. Sweden for example, was the first country to provide additional support for the continuation of organic production beyond the conversion period, and the Swedish and Finnish programmes included support for state advisory services for organic producers. Outside the EU, Switzerland in particular has a long history of political support for organic farming, but policy initiatives have now also taken place in a number of other European countries.

At the EU level two major initiatives addressed organic farming specifically. The first was the EC Regulation 2092/91 defining organic crop production, implemented in 1993. This introduced a legally enforceable common standard for organic crop production, certification and labelling in the EU, which through its provisions for imports from non-EU countries has had world-wide impact on the development of organic standards. The second major initiative at EU level was the inclusion of a direct support programme for conversion and the continuation of organic farming as part of the EU agri-environment programme (EC Reg. 2078/92) that was introduced as part of the CAP reform package from 1993 onwards. The development of the organic sector was further supported through the application of structural measures e.g. schemes to promote investment, training, processing and marketing in the framework of the regionally focused Objective 1, 5a and 5b structural programmes, as well as the funding of research. The regulatory framework for these initiatives and their implementation in the EU member states is reviewed in the following section.

3.1. Legislation Defining Organic Production (EC Regulation 2092/91)

Historically, the situation with respect to institutional involvement in certification has varied considerably across the EU, with private bodies playing the most important role in standards development. In five countries (Austria, Denmark, Spain, Finland and France), governments took the initiative and introduced national, legally enforceable definitions of organic production. In other countries standards were developed and operated by the private sector, in most cases with the involvement of the organic movement, taking regional variations in conditions into account. The growing number of certification marks and labels on organic products on the shelf, including several attempts to present products as ecological, environmentally friendly or natural without making reference to the farming system, contributed to a reduction in consumer confidence in organic products.

In 1991, the EU took up this challenge and introduced legislation defining the use of the term organic production as a means to avoid confusion and fraud, protect the consumer and producer, and hence assist the development of a market for organic food (EC, 1991). The regulation 2092/91 states that a product can only be labelled as organic (or any of the equivalent terms used in other languages e.g. biological, ecological), if it was produced according to rules of organic agricultural production (as laid down in Annex 1), and if the unit of production, processing or packaging has been inspected following the requirements laid down. Since then 19 further regulations and additional legislation amending the original have been passed, e.g. prohibition of the use of genetically modified organisms in organic systems included in EC Reg. 1804/1999, and, after a long period of negotiation, common rules for organic livestock production, while at the same time allowing countries to maintain higher national standards (EC, 1999).

Whether or not the introduction of legislation has achieved the goal of assisting market development is a matter of debate. Geier (1997), the Executive Director of IFOAM (International Federation of Organic Agricultural Movements) sees the regulations as a means to provide more effective and objective control than a system that is essentially responsible for monitoring itself. On the other hand, Rundgren (1997) argued that a further role of standards, that of offering guidance to the producer (and other operators) on how to fulfil their requirements, is sometimes better achieved with private sector standards. Nevertheless, the introduction of a single regulation, as opposed to a variety of standards, is likely to have reduced confusion and increased consumer confidence, although it is difficult to substantiate this at the EU level. At national level some similar initiatives have been successful, for example in Denmark where the state regulation on organic farming is an important reason for high consumer confidence in organically produced food (Willer, 1998). Despite the common legal framework there is some concern about the distortion of competition, particularly in specialist areas of production such as glasshouse production and attempts to further harmonise the standards at international level continue.

For consumers, the development of easily recognisable, common logos may play a more important role than defining legislation alone in improving confidence and reducing confusion. Across Europe, private labels originating from organic producer organisations or the retail sector, some of which are financed by a levy on turnover or organic land area, co-exist with state supported labels. Denmark's single state logo, for example, is recognised by half of Danish consumers, who seem to have high levels of confidence in the system. Switzerland has had a similar experience. On the other hand, the Austrian national logo, which was handled by a semi-private body, is less widely used and not so well recognised by consumers, contrasting markedly with the otherwise rapid development of the sector in Austria (Hofer, 1998). Despite Germany representing one of the largest markets for organic products, a common logo was not introduced until 1999 as a joint initiative between the organic producer umbrella organisation (AGÖL) and the national food-marketing agency (CMA) with relatively limited success. This has now been superseded by the German government's introduction of a state-owned label for all organic products in September 2001 in response to the recent BSE crisis.

Since December 1999, EC Regulation 221/2000 provides for a European logo with the intention of clearly communicating the organic character of the product to consumers, but this is not yet widely used. The EU initiative has the potential to improve the situation considerably, as the same logo can be used across borders, and it is free of charge to any producer or processor of organic products that fulfils the requirements of the regulation across the whole of the European Union. Dabbert et al. (2001) have argued that the logo is likely to be ineffective in its current form, and that major initiatives are required to reform and market it effectively if it is to be useful.

3.2. Direct Payments for Converting and Fully Organic Producers Under the EU Agri-Environment Programme

The EU agri-environment programme (Reg. 2078/92) came into effect in 1993. It offered a range of measures contributing to the achievement of policy objectives concerning agriculture and the environment. Among other options, aid was available for farmers who introduced or continued with organic farming methods, subject to positive effects on the environment (EC, 1992). By 1996, all EU member states except Luxembourg (from 1998) offered organic aid schemes under the regulation. By October 1997, more than 65,000 holdings and nearly 1.3 million ha were supported by organic farming support measures at an annual cost of more than 260 million ECU (Foster & Lampkin, 1999; Lampkin et al., 1999). Organic farming amounted to 3.9% of all agri-environment programme agreements, 5.0% of the total land area and nearly 11% of the total expenditure. The differing shares reflect in part the widespread uptake of baseline agri-environment programmes in France, Austria, Germany and Finland with lower levels of payment per hectare. There were wide variations between countries in terms of uptake and the significance of organic farming support, both relatively and absolutely, within the agri-environment programme (Table 2).

The regulation (supplemented by EC Reg. 746/96) specifies that the payment rates need to be justified in terms of income forgone or additional costs incurred, with the possibility of an additional 20% as an incentive payment. Expected environmental benefits or other costs could not be included in the calculations. The EU covers 50% of the costs of support up to specified co-financing limits; higher EU shares applied in the so-called Objective 1 regions, for which special support policies exist. Payment rates and eligibility conditions varied widely between and within some countries (see Table 3).

Some countries introduced additional environmental requirements in addition to organic standards for farmers to qualify for payments. In Ireland and Finland, participation in the main agri-environment programme was compulsory, for which additional payments were made. In the United Kingdom, additional environmental restrictions, for example the retention of natural habits of high conservation value, have been incorporated into organic production standards at national level. The differing payment rates and conditions have to be recognised as distorting the common European market for organic farming products which EC Regulation 2092/91 was intended to achieve.

Other restrictions on the eligibility conditions are related to the principle of avoiding double payments for the achievement of the same objective under different agri-environment and mainstream measures. Since the introduction of the new Rural Development Regulation under Agenda 2000 (see below), all programmes had to be redesigned although the overall framework for the agri-environmental policies has not been greatly affected by this change. No EU-wide survey regarding the implementation of the new programmes has yet been carried out.

In several countries the types of farms converting under the direct support schemes were skewed towards moderate to low intensity livestock farms, particularly milk production and farms with mixed cropping, and towards marginal areas (Michelsen et al., 2001; Schneeberger et al., 1997; Schulze Pals et al., 1994). Specialist cropping farms (arable and horticulture) as well as intensive pig and poultry producers seemed to be less attracted by the available payment rates. Denmark addressed this by introducing a supplement of 230–266 ECU/ha/year for 3 years for arable farms without milk quota and for pig farms in 1997.

Table 2. Uptake, Public Expenditure and Average Payments for Organic Farming Option under EC Reg. 2078/92 (1997 Data).

Country	Supported organic area (ha)	Supported organic area (% of total agricultural area)	Public expenditure (M ECU)	Organic farming expenditure as percent of total agri-environment payments (%)	Lowest conversion/ organic payment (ECU/ha)	Highest conversion/ organic payment (ECU/ha)	Average payment (conversion and continuing) (ECU/ha)
AT	246 000	7.5	65	12.9	217 (forage)	723 (hortic.)	264
BE	3 401	0.2	1	23.7	180 (cereals)	838 (fruit)	259
DE	229 486	1.3	23	6.0	127 (cereals)	713 (fruit)	101[a]
DK	50 281	1.8	9	58.2	87 (forage)	140 (high N)	188[b]
ES	50 000	0.2	3	3.9	90 (forage)	362 (fruit)	58[c]
FI	89 403	4.2	22	7.6	280 (cereals)	1056 (fruit)	236[d]
FR	41 976	0.1	4	1.4	106 (forage)	711 (fruit)	96[a]
GR	42 600	0.1	4	31.7	182 (cereals)	1217 (fruit)	100
IE	nd	nd	nd	nd	337 (cereals)	398 (hortic.)	nd
IT	308 367	2.0	103	25.6	185 (cereals)	1235 (fruit)	334
LU	0	0	na	na	na	na	na
NL	4 640	0.2	0.3	0.8	226 (cereals)	837 (hortic.)	73[a]
PT	9 938	0.3	1	1.9	217 (cereals)	723 (fruit)	119
SE	205 185	6.5	25	17.0	104 (crops)	254 (livestock)	123
UK	29 127	0.2	1	0.9	20 (LFA)	101 (lowland)	28
EU 15	1 272 064	0.9	261	10.7	181 (cereals)	1208 (fruit)	205

Sources: European Commission and national agricultural administrations; Lampkin et al. (1999).
nd = no data, na = not applicable.
[a] Lower payments for continuing organic farming.
[b] Includes other forms of support.
[c] Estimated.
[d] Excludes payment for main agri-environment protection scheme.

Table 3. Introduction and Implementation of Organic Farming Support under 2078/92 in the Countries of the EU.

Measures	AT	BE	DE	DK	ES	FI	FR	UK	GR	IE	IT	LU	NL	PT	SE
Year of introduction of support															
Previous schemes	91		89	88		90	92				(90)	(92)			89
2078/92 schemes	95	95	94	94	96	95	93	94	96	94	94	98	94	94	95
Eligibility and certification requirements															
Staged conversions allowed	✓	✓		✓	✓	✓	✓	✓	✓		✓	nd	✓	✓	✓
Part farm conversions allowed	✓	✓			✓	(✓)	✓	✓	✓		✓	nd	✓	✓	✓
Existing organic farmers	✓	✓	✓	✓	✓	✓			✓	✓	✓	✓	✓	✓	✓
2092/91 control required	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	
Assistance for control costs	✓			(✓)				✓			(✓)				
Maximum level (per farm)			(✓)	(✓)	✓		(✓)	✓		✓		✓	✓	✓	(✓)
Minimum level (per farm)	✓		✓	✓	✓	✓		✓		✓	(✓)	✓	✓	✓	✓
Permanent pasture excluded			✓		✓			✓			✓		✓	✓	
Set-aside land excluded	✓		✓	✓	nd									nd	✓
Environmental criteria															
Compulsory participation in environmental scheme						✓				✓					
Additional environmental constraints	✓		✓	✓		✓		(✓)	✓			✓			
Information support															
Additional training measures	✓	✓		✓	✓	✓				(✓)	✓			✓	✓
Demonstration and advisory projects		✓	✓	✓		✓	✓	✓			✓		✓	✓	✓

Values in ( ) refer partial/regional implementation.
Source: Lampkin et al. (1999).

No formal evaluations of the environmental impacts of organic farming support as part of the agri-environment programme exist. At a joint conference of the EU Commission and two Austrian ministries in 1999, participants agreed that organic farming in general provided significant environmental benefits and this is supported by the evidence reviewed by Stolze et al. (2000). However, to ensure the environmental benefits of organic farming, a balance between positive incentives (direct support) and restrictive standards (legislation) is required (Anon, 1999). This has not yet resulted in any further development of the European organic standards with regard to environmental impact, apart from a restriction on additional manure inputs which was introduced together with the regulation on organic livestock production (EC Reg. 1804/1999).

3.3. Structural Funds and Processing and Marketing Support

EC Reg. 2078/92 and other conversion support programmes have had a significant impact on the development of the supply base of organic food in most countries where they have been implemented. This poses a challenge for the market, as the establishment of an appropriate structure and new retail outlets is of key importance if the sector is to be able to deal with this expansion and if premium prices are to be maintained (Hamm & Michelsen, 1996). Some support has therefore focused on assistance for the development of the market, but few countries saw the need to integrate market support programmes with producer support schemes.

At EU level, one of the established priorities for the application of EC Regulation 866/90 on improving the processing and marketing conditions for agricultural products (EC, 1990) is investment relating to organic farming products. In eight countries (Austria, Germany, Spain, Finland, Italy, the Netherlands, Sweden and the United Kingdom), organic food related activities have benefited under this regulation (Lampkin et al., 1999). Restrictive eligibility requirements were identified as one of the barriers to greater uptake of EU support. In Germany and Denmark, the regulation resulted in the introduction of a national programme to support the marketing of products 'produced according to specific production rules' that takes the specific requirements of the organic sector, such as reduced turnover and smaller numbers of members of organic producer groups, into account. Other countries, for example Austria, have implemented national or regional grants and/or programmes supporting marketing and processing, under which organic enterprises can and have received funding.

Experiences in Denmark suggest that a more market-oriented approach to organic aid schemes can promote the development of a diverse marketing structure, provide help in entering into mainstream marketing, and help overcome problems such as discontinuity of supply and lack of widespread distribution.

In view of the potential contribution to structural adaptation of agriculture and rural development, organic farming projects have also received support under Objectives 1 and 5b of the EU Structural Funds and the EU LEADER programmes in 11 states (Lampkin et al., 1999). These projects cover a variety of activities, including direct marketing, promotion of regional products, research, technical advice and training. Some regional development schemes include support for marketing and processing activities in the organic sector, mainly aimed at small-scale projects. Such schemes have been particularly successful in Germany in helping the development of regional marketing networks, overcoming the problems of a small organic sector and encouraging the entry of new operators. The impact of grant aid on the organic sector and consequently the development of the region can be significant as evaluations of the Irish Objective 1 programme have shown (Fitzpatrick, 1997).

3.4. Information-Related Policy Measures

Knowledge and information services for organic farming have received little specific attention from policy makers and researchers. This is surprising, as, like other forms of sustainable agriculture, one of the key features of organic farming is that it is knowledge-intensive, as inputs are replaced with management of the agro-ecosystem for which information is of prime importance (Lockeretz, 1991). Studies on farmer attitudes to conversion give an indication that, in many areas, access to information is a problem for interested farmers, as well as a lack of specific technical knowledge indicating a need for further research (Chadwick & McGregor, 1991; Vogtmann et al., 1993; Midmore et al., 2001).

In the EU, the situation with regards to organic information and advisory services varies between countries, with widely available, subsidised advice for organic and in-conversion producers through the main agricultural extension services (e.g. in Finland, Denmark and Germany) or through private bodies (e.g. in the United Kingdom), to countries with an almost fully commercial basis for any professional advice (e.g. Spain, Greece, Ireland and the Netherlands). An increasing involvement of general agricultural extension services in organic agriculture is likely to increase the availability of introductory information for interested conventional producers. Organic producers, however, are sceptical whether mainstream agricultural advisors can provide information that is specifically tailored to their systems and would generally prefer specialist organic advisory services (Fersterer & Gruber, 1998; Gengenbach, 1996).

Most support for organic information and advisory services has developed as part of national initiatives, such as general grants to governmental extension and advisory services and for training and research. However, specific organic conversion information programmes in Sweden and in the United Kingdom have proved very popular, confirming that the proportion of agricultural producers interested in considering organic conversion is far greater than the number of farmers who have actually converted (SA, 1999; Burton et al., 1999; Midmore et al., 2001).

At the EU level, provision for farmers' training was made in Regulation 2078/92 (the agri-environment programme, see above), which was used by some member states to develop information services, training programmes and demonstration farm networks for organic production. The topic "organic farming" was also included in the work programmes for several EU-wide research programmes for agriculture (CAMAR, AIR, FAIR) and up to 1997 the EU had funded a total of ten research projects relating to organic farming (Lampkin et al., 1999). Some further research has been commissioned under the fifth Framework Programme.

In all EU member states organic farmers and growers remain a very important source of information. In some cases this role has been recognised through grants to organic producer organisations. Regional organic producer groups facilitate the sharing of experience among organic farmers, act as a focal point for regional market development and give social support to producers. The organisation of such groups is mostly voluntary, apart from four countries (Austria, Belgium, Italy and the UK) where the co-ordination of regional groups received support under the EU Objective 5b Structural Funds.

3.5. National Integration of Policies to Support Organic Farming

The policy measures supporting organic farming that were reviewed so far can be broadly characterised as push or pull measures, focusing either on the supply side (push), for example the agri-environment measures and to a lesser extent advisory, training and research programmes, or on the demand side (pull), for example the legislation defining organic production, marketing and processing grants and, to a lesser extent, the structural measures.

In most countries the implementation of these different measures was not co-ordinated with each other or with mainstream agricultural policy. Within the push measures, for example, the interaction between the organic farming option and other agri-environment measures was frequently not considered, in particular when other competitive schemes, such as support for reduced input use in agricultural production, were introduced. This is illustrated by the substantial number of producers that were supported under the organic farming option in the regions of Tyrol and Salzburg in Austria and left the scheme to take up alternative, low input options with fewer restrictions (Kirner, 1999; Michelsen et al., 2001). Similarly, the interaction between producer support under the agri-environment programme (EC Regulation 2078/92) and the mainstream CAP measures (e.g. arable area payments, livestock premiums and quotas) received relatively little attention from policy makers, despite the potential conflict between them (Lampkin et al., 1999).

Most problems, however, were observed through a lack of integration between push and pull measures leading to imbalances in the market place. For example, the German organic food markets at the end of the 1980s could not absorb the substantial increase in supply that followed the introduction of conversion aid payments under the German extensification programme. This was due to the fragmentation into a number of producer organisations with their specific trade marks and a focus on direct marketing and specialist organic food shops. At first, very few specific measures to support market development were introduced and, despite a small increase in demand, this oversupply led to a sharp decline in farm-gate prices and marketing problems for many existing organic and converting producers. A substantial proportion of organic products had to be sold in the conventional market without any organic premiums (Hamm & Michelsen, 1996). Similarly, lack of marketing opportunity for mainly livestock products is believed to be one of the reasons for some producers deciding to re-convert and a major obstacle for the further uptake of organic farming methods in Austria (Kirner, 1999; Kirner & Schneeberger, 1999). This contrasts markedly with the Danish experience with a more market led approach to organic farming support, although Denmark has also experienced policy-induced imbalances between supply and demand, in particular in the milk sector. Across the EU, the larger uptake of organic production among livestock producers compared to arable production has led to some supply problems with organic cereals for feeds.

To overcome these difficulties, several countries developed a more integrated approach to supporting the organic sector. All the Nordic countries, the Netherlands, France and Germany have developed integrated policy programmes or so-called "action plans" for the future development of the organic sector. In addition to a clear target for expansion of the producer base, these action plans integrate a variety of policy measures to achieve their goals, such as support payments for producers, harmonisation of certification procedures, market support, as well as support for advisory services, training and research and development. Some also include a detailed analysis of the state of the organic sector in the country, targeting specific measures to address particular problems identified.

As part of the actions plans, Sweden, Austria and some *Bundesländer* in Germany were officially committed to and close to achieving, a target of 10% of agricultural land managed organically by the year 2000. France, the Netherlands and some other Nordic countries (Denmark, Finland) have set lower targets of 3 and 5% respectively. Wales was the first region in the United Kingdom to announce a target of 10% by 2005 in 1999 and Germany has announced a new national target of 20% organic production by 2010 (Kuenast, 2001). Some non-EU countries have also set targets, for example Switzerland has a target of 90–95% of farms producing to integrated farming standards by 2002, with the remainder organic, and some cantons are committed to 10% organic. Most other EU countries have no specific targets for the development of organic farming.

4. TOWARDS A EUROPEAN ACTION PLAN FOR ORGANIC FARMING

In this paper, we have adopted an approach to policy evaluation that takes the actual and proclaimed objectives of politicians, such as maintaining farm incomes, minimising environmental impact of agriculture, and enhancing rural development, as a starting point for analysis (Dabbert, 2000), with the main aim of assessing whether and how organic farming can contribute to such existing policy objectives. Section 2 concluded that supporting organic farming can contribute to some current goals of agricultural policy in Europe, such as to the achievement of agri-environmental benefits and surplus reduction, and that the market for organic products can support these, although many questions, for example a clear assessment of the benefit for rural development, remain the subject of further research. However, in focusing the review on support policies for organic farming, our analysis does not allow any judgement to be made on whether alternative policy measures would be better placed to achieve the same goals.

The policy review presented in Section 3 illustrates that the commitment of individual countries in the EU to supporting organic farming varies considerably despite a common European policy framework. This may be one explanation for the significant difference in the uptake of organic farming between the member states (see Fig. 1). A range of factors influencing individual producers' decisions for or against organic conversion have been suggested, including: access to markets and consumer demand for organic produce; changes in the relative economic profitability of organic farming compared to general agriculture; the availability of competitive schemes; public opinion and opinions in the farming community; and the availability of knowledge and information (Michelsen et al., 2001; Padel, 2001).

Periods of stagnation or declining numbers of organic producers in several countries even when support policies continued to be in place illustrate that differences in implementation of policy support alone cannot fully explain the variation. Some of the factors identified as influencing the growth of the sector are likely to be outside the range that agricultural policy can influence, such as public opinion and consumer demand. Other factors, however, such as the availability of competitive agri-environmental schemes and a lack of access to marketing support are clearly related to a lack of integration between policy measures highlighted in the previous section.

At EU level, the re-orientation of agricultural policy under Agenda 2000 included a shift in agricultural support away from price and production-based support programmes towards the development of integrated support for rural communities. As part of this, all policies in the area of agri-environment, rural development and structural support were consolidated into a single Rural Development Regulation (EC Regulation 1257/1999) and all member states are now required to produce customised rural development plans for their regions. In addressing all issues relating to rural development in one single regulation, the new policy framework under Agenda 2000 has some parallels to the 'action plans' for the development of organic farming in individual EU member states.

With the combination of aiming for a sustainable model of agriculture, and the encouragement of local production, processing and consumption patterns and marketing networks, organic farming could help meeting many of the goals of regional development and lead to an increase in the 'economic value' of a region (Vogtmann, 1996). In an exploratory search, Pugliese (2000) found strong convergence between organic farming and the four cornerstones of sustainable rural development, defined by rural sociologists as innovation, conservation, participation and integration, whereby organic farming systems could contribute to all aspects. Research funded by the Commission of the European Communities and currently under way includes the development of a conceptual framework on how the contribution of organic marketing initiatives to rural development can be evaluated in a multi-perspective approach (see note 4 below).

The potential contribution to rural development was also recognised by the EU commissioner for agriculture Franz Fischler, stating that organic farming may play a potential role in some regions as part of an integrated approach towards sustainable rural development, whereby agri-environmental measures at the heart of the strategy could be supplement by rural development measures such as farm investment and start-up support for processing and marketing units (Fischler, 1999). However, although providing a framework, the Rural Development Regulation does not specifically address the problems of the European organic farming sector and market for organic food.

This review has highlighted that differences in the implementation of aid programmes and in the interpretation of legislation defining organic production represent distortions in the level playing field for, and are obstacles to the development of, a common European market for organic food. A more integrated approach to policy development in support of organic farming at European level, co-ordinating push and pull mechanisms as well as policy making across the countries and regions, could be achieved through a European action plan for organic farming (Dabbert et al., 2001). A first step in developing such an action plan should be to identify a coherent description and understanding of the situation of organic farming in Europe at the outset and the barriers to further expansion, identifying carefully strategic decisions that have to be taken, such as the reasons why policy-makers want to support organic farming, the weight and regional scope and focus of support policies, e.g. preferences for support in marginal areas or targeting particular farm types and the importance of protectionist elements in organic farming policy. Answers to such questions are important in choosing an effective policy mix.

The following policies should be part of a European action plan:

- Informing the consumer, especially developing a unified approach to widely recognised common logo based on EU regulation 2092/91 and subsequent legislation.
- Improving the functioning of the organic chain: production, processing, distribution within the supply chain with major emphasis on improving information, education, technology development, research and extension for organic farming and its process chain.
- Supporting organic farmers financially as remuneration for the production of public goods.
- Reviewing related policies with direct influence on organic farming such as the general measures of the Common Agricultural Policy or tax laws and others, in particular removing production constraints such as set-aside aimed originally at conventional producers.
- Supporting a "creative" conflict between conventional and organic farmers and building as much as possible a consensus on the long-term objectives whilst respecting the integrity of organic farming.

A high priority should be given to measures dealing with improving the information on organic farming to consumers. It is recommended to closely monitor any realisation of an action plan and to adapt the plan accordingly. In order to turn the development of the plan into a catalytic process for the organic sector it is important to involve the stakeholders in a systematic way.

A call for an EU wide "action plan" was supported by 9 EU and 4 non-EU countries whose agriculture ministers attended the European conference "Organic Food and Farming – Towards Partnership and Action in Europe" in May 2001 in Copenhagen (MFAF, 2001), and subsequently by the EU Council of Agriculture Ministers meeting in June 2001.

The process of developing an action plan would require the willingness to pursue new approaches to policy development on the part of the European Commission, based on wide consultation and partnership with stakeholders in the private sector. On the side of the organic farming movement in Europe, this implies the need to recognise and accept the changed role from a social movement to an object of agricultural policy that took place during the 1990s (Dabbert et al., 2001). To continue this process will not be without problems and further growing pains. It requires from all sides the clear commitment to support the further development of the organic farming sector in Europe and an active and constructive engagement in the political process.

NOTES

1. 1 Euro = $0.90.
2. ECU = European Currency Unit, an accounting unit pre-dating the introduction of the Euro, where 1 ECU = 1 Euro.
3. For details of the project see http://www.uni-hohenheim.de/ ~ i410a/eu_org/Fair3_Index.htm
4. OMIaRD – Organic Marketing Initiatives and Rural Development (QLK5-2000-01124). http://www.irs.aber.ac.uk/omiard

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9. DOES IT MAKE SENSE TO BUY LOCALLY PRODUCED ORGANIC PRODUCTS?

David Vanzetti and Els Wynen

ABSTRACT

Enthusiasts of the organically grown food industry often espouse a preference for produce grown in the local region, and suggest that consumers should buy locally produced organic products. One reason consumers buy organic products is to improve the environment. There is a perception that transporting foods long distances is wasteful, in part because transport costs are not appropriately priced to include all externalities. Does this make sense?

The focus of this paper is to examine conceptually how trade can contribute to a more environmentally-sound way of supplying agricultural products to consumers, even when transport costs are adequately taken into account. An example from the international wheat trade illustrates this point.

1. INTRODUCTION

There are many and varied reasons for consumers to prefer organically-grown food. Meier-Ploeger and Vogtman (1996, p. 176) mention appearance, technological quality (protein or starch content) and biological quality (taste, freshness, absence of toxic substances) as characteristics of interest to

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consumers of organic products in former years. More recently '... ethical criteria such as the environmental, social and political dimensions of food production, processing and packaging' have become more important.

Considerations for the environment, and the issue of local consumption or transport of organically-grown produce is the focus in this paper. Preference for produce grown in the local region is espoused by some on the grounds of better environmental management (see, for example, Marsh and Runsten (1997), who quote Wilkins (1995)). Transport costs are seen as an unnecessary waste, and it is considered that non-renewable resources should be used sparingly, thus providing for future generations. The regulatory body overseeing organic standards, the International Federation for Organic Agricultural Movements (IFOAM) tends to favour locally-grown produce, although its position on this is nebulous, merely advocating the use of 'as far as is possible, renewable resources in locally organised agricultural systems' (IFOAM, 2000).

In a survey of ethical trading, Browne et al. (2000, p. 76) note that several respondents within the organic movement were concerned about the 'negative environmental impact of transporting food over long distances from developing countries'. A representative statement of many in the organic movement is forcefully put by Lang (1996, p. 200), who maintains that

> Food travels an increasing distance between producer and final consumer. Some – most – of this travel is ludicrous but it makes financial sense because the cost, in energy and money, is externalised onto the environment. Cheap beef or rice going from the USA to Japan relies upon cheap oil, a non-renewable resource.

The purpose of this paper is to explore the contribution of transport costs to overall resource use. While moving goods long distances obviously increases transport costs, offsetting savings can be gained from producing goods with the use of fewer resources in distant locations. If transport costs can be shown to be sufficiently small, the policy of 'buying locally' can be shown not to be sensible, at least not for the reasons commonly espoused. However, there may be more sensible reasons for buying locally-produced organic products if consumers place a sufficiently high weight on local as opposed to global environmental benefits. The links between international trade and environmental issues are explored in this paper.

In the next section, the nature of the gains from trade in general is discussed. The role of transport and its use of, possibly underpriced, non-renewable resources is discussed next. The link between environmental and trade policy is then examined, and finally attention is given to the policy implications for organic agriculture.

It is worth at this juncture defining the term 'locally produced', as this means different things to different people. Some believe it refers to consumers being in touch with producers, while others interpret it as meaning supplies are sourced from within the state or province. More generally it has national connotations. National borders are irrelevant for transport costs, leading to the ludicrous situation where it is deemed acceptable to transport vegetables from Wales to Scotland but not over the border from France to Germany. Some see the European Union as a single entity, and regard transport within Europe as acceptable but trade with non-members as somehow undesirable. In the following discussion we regard local production as being traded nationally rather than internationally. There are three reasons for this. First, there is a nationalistic perspective, with consumers being extolled to 'buy local' because 'the money stays at home', providing jobs for local people. Since most countries have tax policies that redistribute incomes from one region to another, most citizens are more agreeable to sharing with their fellow citizens than foreigners. The second reason is that policies can be applied at the national levels to encourage consumers to buy locally-produced products. Restrictions of trade at the provincial or state levels are less common. Finally, a pragmatic reason is that data on international trade is more readily available than regional data. Nonetheless, much of the analysis and reasoning presented in this paper can be applied to any level of aggregation, be it village, local government area, province, nation or trade bloc. The key issue is the cost of transportation versus the benefits of producing something more efficiently at a more remote location.

Unfortunately, estimates of the volumes and values of organic production and trade are unavailable, and thus this paper is limited to a conceptual discussion. However, data relating to trade in organic Australian wheat is used to illustrate the potential gains and losses.

2. GAINS FROM TRADE: A CONCEPTUAL ANALYSIS

What is the source of the gains from trade? Different countries are endowed with differing levels of various resources, such as land, labour, capital, minerals, water and many other factors. With respect to agriculture, the abundance and quality of the soil and the prevailing climatic conditions determine the agricultural potential. These factors influence the costs of production, and hence lead to different agricultural output prices between countries. Of importance in determining trade flows are the relative, not the absolute, costs of production. Countries that could produce everything more cheaply will find it in their interests to specialise at those products at which they have the greatest comparative advantage. Consider this illustrative analogy.

> A farmer does off-farm contract work planting trees for $500 a day, and employs a labourer to do the milking for $150. Furthermore, while the labourer can milk 20 cows per hour, the farmer can milk 25 cows. Our farmer is not only a better tree-planter, but also a better milker than his employee. Should he spend some of his time milking. The answer is clearly negative. He could earn $180 per day milking (assuming earnings relate to output), but only by giving up $500 worth of tree planting. $500 is the opportunity cost of a day's milking.

Here, somewhat oversimplified, is the basis of trade: specialisation. A further illustration is presented in Fig. 1. The key point is that an efficient use of resources allows more to be produced – and hence consumed – at the same level of input use. Alternatively, the same amount could be produced and consumed with the use of fewer inputs. As the depletion of scarce resources and environmental pollution tends to be related to input use (rather than the level of output), it is tempting to conclude that removing impediments to trade is unambiguously beneficial to the global environment. However, such a sweeping conclusion would be premature.

3. TRANSPORT COSTS

One barrier to trade is transport. The significance of transport costs depends on the value to weight ratio of the product. Transport costs must be less than the difference in relative prices of the goods traded. If potatoes cost $100 per tonne in one country, and $110 in another, trade will not be viable if transport costs exceed $10 per tonne. Freight costs are most likely to exceed price differences on low value (per kilogram) products, such as turnips or potatoes, or on those that are difficult to store and transport, such as fresh milk, eggs, livestock or some vegetables. This implies, ironically, that there may be greater scope to transport organic produce over international borders, due to its greater value than conventional produce.

However, the price of the transport may not reflect the true costs to society. Most forms of transport cause some pollution that is not paid for by the users of the transport system. Noise, air pollution and road damage are some obvious examples. These objections apply more to land transport than sea freight. Transport by sea has resulted in some noteworthy disasters, but these generally involve the transport of oil itself, rather than goods produced in using the oil. The aggregated level of pollution associated with sea transport of commodities and manufactured goods is minimal. A further argument is that transport costs are wasteful, as transport is dependent on a non-renewable resource, oil. Implicit in this is the view that oil is underpriced. This view has some validity, and is examined below.

Assume two countries produce only two goods, and can trade only with each other. Suppose Country A can produce 10 tonnes of wheat or 4 tonnes of rice, or any combination, using a given quantity of resources. This is shown in Table 1.

Table 1 Hypothetical Production Possibilities.

	Wheat	Rice	Relative prices
Country A	10	4	2.5
Country B	20	14	1.43
Relative Production Cost A:B	2:1	3.5:1	

Country B can produce 20 tonnes of wheat or 14 tonnes of rice using a similar quantity of resources. Thus, in terms of the resources used, Country B is more efficient at producing both wheat and rice. Intuition might suggest that it should produce both its own wheat and rice, and not trade with Country A. However, this is misleading. B can produce rice 3.5 times as cheaply as A, and wheat twice as cheaply. It therefore has a relative advantage in producing rice.

Country A could produce an additional 2.5 tonnes of wheat by forsaking 1 tonne of rice, whereas Country B could produce that additional tonne of rice using the resources released from giving up 1.43 tonnes of wheat. Thus, the same amount of rice and (2.5–1.43) 1.07 extra tonne of wheat are produced with the same resources. This is illustrated in Table 2. In the pre-trade example, each country uses half its production capacity in producing wheat, and the other half in producing rice, totalling 15 and 9 tonnes of wheat and rice, respectively. In the post-trade scenario, Country A produces one tonne less of rice, and 2.5 tonnes more of wheat. Country B produces one tonne of rice more, thereby dropping 1.43 tonnes in wheat production. This arrangement yields 16.07 and 9 tonnes of wheat and rice, respectively.

The price at which the goods are exchanged and the levels of production and consumption in each country depend on the nature of consumer preferences in each country (not specified in this example). It is clearly to both countries' mutual advantage to specialise in this fashion.

Table 2 Impact of Trade on Production.

	Wheat	Rice
Pre-trade		
Country A	5	2
Country B	10	7
Total	15	9
Post-trade		
Country A	7.5	1
Country B	8.57	8
Total	16.07	9

After trade has opened up, the relative prices will be the same in both countries, assuming no transport costs, somewhere between the relative prices 2.5 and 1.43. However, trade has enabled total production and consumption to increase from the same volume of resources. Clearly, trade could also facilitate a given level of consumption with the use of fewer resources.

Fig. 1. The Gains from Trade.

4. OPTIMAL RESOURCE USE

Determining the optimal use of finite resources is a highly complex mathematical problem, and subject to considerable uncertainty. In theory at least, the interplay of market forces will provide an efficient, optimal allocation of resources over time, in the sense of providing the greatest benefits (leaving aside problems of measurement and distribution of gains). As a resource is depleted, its price tends to rise, reflecting its scarcity value. Of importance here is the likelihood of developing suitable substitutes for non-renewable resources. As the resource dwindles, the rising price encourages the search for substitutes. In the case of oil, it is difficult to think of a use for which an alternative is not available, albeit that these alternatives are currently much more expensive with current technology (for example, solar power or fuel cells).

However, the conclusion that an optimal use is made of the non-renewable resource oil can best be seen as a benchmark, as there are imperfections in the market. The existence of monopolies (which supply less and charge more than in a competitive market) and uncertainty regarding the available resources tend to underutilisation; a number of other factors lead to overexploitation.

The main factor leading to over-exploitation relates to the preference for individuals to consume now rather than postponing consumption until later. Most of us would prefer to receive $100 now than at the end of the year. This preference is reflected in the discount rate, which can be thought of as the opposite of a compound rate of interest, and is used to compare future costs and benefits with those of the present. This is especially important when environmental issues are considered, because current actions have effects stretching well into the future. A higher (discount) rate implies there is a preference for consumption to be brought forward. This rate may be lower for society as a whole than for individuals. That is, the preference of the society as a whole might be to consume less now to have more left later. Hence, the rate of resource use will be too fast if private individuals are making decisions concerning the rate of use. One factor influencing the social discount rate is the need to provide for future generations.

A second factor leading to overuse of energy resources is the underpricing of the pollution and other externalities associated with the use of energy. Externalities include the costs of accidents, congestion, noise and local and global pollution. Although there are difficulties in measuring these costs, and hence data should be used with care, an OECD report suggests costs in Germany of around 25.8 Euro per tonne per 1000 kilometre for road transport, 3.7 Euro for rail and 1.8 Euro for waterways (Quinet, 1999, p. 28). This implies that the shipping of bulk commodities or processed food items by sea or rail is relatively pollution free. Another estimate suggests full internalisation of transport-related externalities would raise costs to end-users (drivers, passengers or distributors) by 15–30% (OECD, 1999, p. 16). These costs are substantial, but the bulk of them (attributable to accidents, noise and congestion) occur in moving trucks in and out of cities, not between cities. To the extent that there is an environmental problem associated with moving food and other goods around the country or around the world, this is best addressed by encouraging the greater use of rail rather than road. This already happens to some extent. European drivers pay more than double world prices for fuel.

5. INTERGENERATIONAL EQUITY

Intergenerational transfers are largely an equity problem. One view is that all generations are linked through concern for one's children, who will in turn care about the fate of their own offspring. Thus, future requirements are taken care of by these inter-generational concerns. Furthermore, future generations will take care of themselves, just as this generation has, through the development of new technologies, and the substitution of capital for scarce resources. This argument may be valid for relatively short periods, such as one or two generations, but appears to have less weight when centuries are considered, where the link between generations is more tenuous, and is less convincing where irreversible decisions (such as agricultural production methods causing soil degradation) are made. Decision makers need some means of weighing current and future benefits. A discount rate provides the means. Benefits are worth postponing if they increase faster than the discount rate, which is somewhat akin to the long term rate of interest. Some commentators favour a discount rate for society that is below the private rate, because future generations are not present to represent their interests. A lower discount rate encourages investments with long-term benefits, such as soil conservation, and discourages activities with long-term costs, such as the application of fertiliser that may pollute water supplies 30 years hence. This raises the dilemma of assessing investment decisions against two different benchmarks, the social and the private discount rates. There is also the problem of deciding what the social discount rate should be (see Fisher (1981) for a discussion of these issues.)

Is oil under or overutilised? Is the price right? What would be the correct price if the social discount rate, intergenerational transfers, pollution, the potential for substituting alternatives in the future and other relevant factors were taken into account? One can only guess at this. It is tempting to presume that the price would rise, implying that, at present, the resource is likely to be overutilised. However, a resilient feature of almost all commodity prices is a long-term decline, occasionally interspersed with sudden price spikes. Crude oil prices have exhibited a flat trend in inflation-adjusted terms since 1870. Between 1948 and 1957 prices fluctuated between US$14-$16 in inflation adjusted (1996) dollars. Since the Gulf War in 1991 prices have generally been below their long term average of US$19.27, reaching $12 in 1998 following the Asia crisis before recovering to around US$20 per barrel in 1999 in response to the boom in the U.S. economy (WTRG Economics, 1999). Prices rose to over $30 per barrel in early 2000 following OPEC output restrictions before falling back in 2001. With policies aimed at reducing global warming likely to lead to a shift away from carbon fuels, there is little evidence as yet that resource depletion will lead to a long-term sustained rise in real oil prices.

However, if the price of oil would rise substantially and be sustained, what effect would this have on the international food trade? We look at the world wheat trade as an example.

6. AN EXAMPLE: THE INTERNATIONAL WHEAT TRADE

In contrast to the production of many goods and services, agricultural production is particularly influenced by climate and soil conditions. Thus, a limited number of countries are the most efficient at producing any given crop, such as wheat or, more strikingly, bananas. Although production is localised, consumption is not, and hence there is scope for international trade.

Freight rates for conventional bulk wheat from the U.S. Gulf to Rotterdam were estimated to amount to about US$14 per tonne in the mid-1980s (IWC, 1989). Later estimates indicate that average cost of freight has fallen on this route since the beginning of the 1980s and in 1995 amounted to 7% (US$8–10) of landed costs (Ocean Shipping Consultants Ltd, 1996, p. 30). Fuel costs amount to about 20% of the total shipping costs. A doubling of fuel costs would therefore increase freight costs by US$2 per tonne, that is, an increase of 20% on US$10. This is an insignificant amount compared with the cost of production differences for products like wheat, where the world price for conventional product of around US$140 per tonne (ABARE, 1999) compares with a European Community price of approximately 50% more. For example, in 1998 the average price per tonne of bread-making wheat in the European Union ranged from ECU108 in The Netherlands to ECU151 in the United Kingdom (European Commission, 1999). In addition, farmers receive ECU55 per tonne as compensatory payments. If the ECU achieved parity with the U.S. dollar, this means that production costs in the EC would be around US$205 per tonne, while farmers in some countries can profitably produce one tonne for US$140.

Where does the extra cost go? It eventually returns to the owners of primary factors, land, capital, labour and natural resources (such as oil). Differences in land prices account for most of the difference between high and low cost countries, reflecting the scarcity of land in some high-cost countries. However, in some countries, such as Norway, extra functions such as the drying of grain may require additional energy.

Unfortunately, data relating to energy use in organic production in different countries are not adequate to draw conclusions. It is not possible to determine, for example, whether the production of a tonne of organic wheat in Germany uses more or less energy than a tonne produced in Argentina. This is because energy use varies tremendously depending on the situation, and there is no standard means for measuring energy use. (See Stolze et al., 2000, p. 69 for a discussion of some of the difficulties of measuring energy use).

A simplified analysis based on these figures suggests that by producing one additional tonne of conventional wheat in the European Union and importing one less, production costs in this region rise by around US$205. However, total production costs fall in the exporting regions, by around $140, and transport costs fall by around $15. The net global loss is thus the difference in prices minus the transport costs, that is, US$50 in this example.

What is the situation with organic wheat? At the present level of production, the transport costs for organic grain are greater than for conventionally-grown wheat – as it is shipped in containers rather than in bulk – but the value of the product is also greater. The cost of shipping a container of wheat from Australia to Europe is around US$40 per tonne (Ian Diamond, trader, Organic Connections, personal communication, May 1999). With costs of production at around US$140–150 per tonne, on a par with conventional wheat, the landed costs (including transport) of Australian organic wheat in Rotterdam are therefore under US$200. In comparison, Danish producers of organic winter wheat received Dkk1859 in 1995 (Wynen, 1998, p. 61), and German farmers around DM860 in 1992/93 (Padel & Zerger, 1994, p. 92). These figures amount to US$230–260 depending on the exchange rate. Year-to-year price changes plus various policy impediments to trade make comparisons difficult, but nonetheless, it is clear that higher prices are required to induce European organic farmers to deliver grain to the European market than are required to induce Australian (or U.S. or Canadian) farmers to do so.

The relative yields for organic compared to conventional produce tend to be higher in less-intensive agricultural systems such as those used in Argentina, Australia or the United States which are not so reliant on fertilisers and pesticides. This implies that there is a tendency for price differences between countries to be greater for organic than conventional products, providing greater incentive for trade.

The case for trade is obviously greater for products, such as bananas, that are expensive to grow in Europe. Meat and dairy products also have low transport costs compared with the cost of production. Meier-Ploeger et al. (1996, p. 212) indicate that these products are quite energy intensive in production, whereas for vegetables, fruit, sugar, beverages and grain products, the bulk of the energy use and cost between farmer and consumer goes into processing. There is a less strong case for trading these goods on the basis of minimising energy costs. However, processing is often labour intensive, and hence there is a case for developing countries processing their own goods, such as coffee, rather than exporting the raw product for processing in developed countries.

It should also be noted that, although the cost of transporting organic produce may be higher at present than conventional produce, the environmental costs of the transport are similar. The extra expense is taken up in storage, handling, packaging, insurance and commission rather than fuel. These expenses are likely to diminish as organic trade increases.

Shipping costs are likely to be a less significant proportion of the final price for products of higher value. A tenfold increase in fuel prices is unlikely to make trade in wheat totally unprofitable, although it would certainly diminish both international and national trade in wheat and other products.

In summary, transport costs provides little justification for local consumption. Even assuming that present transport costs do not reflect the true costs, total resource use for the production and transport of a good can be lower when transported internationally than produced and consumed locally.

7. IMPLICATIONS AND CONCLUDING COMMENTS

Specialisation of production via international trade provides for substantial increases in production at a lower resource use. This is true not only for wheat but also for sugar, dairy products and beef, products for which European prices are up to two to three times world levels. Some products in Japan are 8 or 10 times world levels. This implies that resources that are required to produce these products in these areas could much more effectively be directed elsewhere. Furthermore, the over-intensive use of agricultural land is likely to lead to environmental problems that would not occur to such an extent in less intensive production systems in countries where farmers receive unsubsidised world prices.

Purchases of locally-produced products at higher prices accentuate these problems, to the detriment of people in all countries. Substantial rises in fuel costs for transport would be necessary to eliminate the potential gains from locating production more appropriately.

However, there is one further consideration that may favour the consumption of locally-produced organic products. Agriculture is environmentally degrading in production, rather than consumption (in contrast to coal, for example). A consumer concerned with the local environment should buy products grown elsewhere, whereas a globally minded consumer should buy products grown where the resource use is least. Organic production avoids some of the pollution impacts associated with conventional production. Hence, buying locally-produced organic products rather than locally-produced conventional products would appear to be beneficial to the local environment. However, if consumers in all countries think along these lines the outcome is less favourable, as resources aren't used as efficiently as possible.

Rather than espouse purchases of locally-produced products, a more fruitful approach may be to encourage governments to play a more active role by initiating polluter-pays policies. A tax on pesticide and fertiliser use is one such example in agriculture. Indeed, several countries in Europe have taken this approach. Such policies are likely to prove beneficial to European organic producers, consumers and environmentalists and to producers in developing countries. However, reducing or removing subsidies that led to overproduction in the first place would also bring about substantial environmental benefits.

There may be sound social, political and environmental reasons to prefer locally-produced goods and there may also be economic reasons not discussed here. Underpriced transport costs appear not to be an adequate justification. Consumers should bear in mind that, where locally-produced goods use more resources to be produced, global environmental benefits may be foregone.

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10. THE IMPACTS OF ALLOCATION STRATEGIES FOR SPATIALLY REGULATED CHEMICAL USE

Lori Lynch and Janet Carpenter

ABSTRACT

Spatial regulations can restrict chemical use more efficiently by linking local benefits to local costs. California has instituted such a spatially based regulation of an agricultural fumigant to meet air quality standards. We examine the implications of alternative allocation mechanisms: allocation of use based on a first come, first served basis; on quotas linked to historical use; and on the highest-value use. Although there are distributional impacts by crop, the overall change in aggregate value from using a highest-value use mechanism rather than a first come, first served approach is estimated to be less than $9 *million of a total potential regulatory cost of* $65 *million.*

Spatially based environmental regulations recognize that different areas or micro regions have different costs and benefits of maintaining environmental quality. Instead of imposing uniform regulations for the entire country or for an entire state, spatially based regulations link the value of the improved environmental quality to the cost of achieving it for a particular area. For example, regulatory agencies may impose stricter pollution abatement requirements in areas with higher pollution levels. Cost differences could be

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based on an area's ability to absorb or dissipate pollutants or on the type of crop or products produced. Alternatively, benefits may be higher due to the value of the environmental resource or amenities in a region such as a drinking water source. Thus, regulations may be set on the value of the environmental resource to be protected. Both the Clean Air Act and the Clean Water Act incorporated spatial dimensions to achieve minimum standards. Under the Clean Air Act, more polluted areas such as the Los Angeles basin demand higher emission reductions from the sources of air quality contaminants to achieve the mandated uniform air quality standards. Under the Clean Water Act, states impose limits on total daily loads entering certain water bodies that depend on the ability of these water bodies to dilute the pollutants and the value of the water resource to be protected.

As a rule, pesticide regulations have not incorporated a direct spatial component. We found few regulations that had such a component. The exceptions include the Endangered Species Act, which proposes voluntary limits on chemical use in areas where endangered species may be affected, the limitation on use of twenty-three pesticides of "toxicological concern," which can be used only as part of an approved integrated pest management (IPM) program in an area of Massachusetts, and the limits on the herbicide atrazine, which is restricted in certain areas of Iowa and Wisconsin where the likelihood of the pesticide leaching into ground and surface water is high. This lack of a direct spatial component in other pesticide regulations exists despite research demonstrating the regional differences in impacts of chemical regulations and stressing the need for disaggregated analyses (Lichtenberg, Parker & Zilberman, 1988; Carpenter, Gianessi & Lynch 2000; VanSickle, Brewster & Spreen 2000). However, in 1994 California introduced a spatially based pesticide regulation that limits the amount of 1,3-dichloropropene (1,3-D), a fumigant pesticide, that may be used in a 36-square-mile area. The regulation seeks to ensure that ambient air concentrations of the pesticide emissions do not exceed air quality standards. This pesticide limit will be binding only in areas of relatively concentrated use. Yet, given that more than one farmer contributes to the adverse environmental effect, some method of allocating the right to use the pesticide must be determined for areas where the regulation is binding. Our paper examines the impacts of different allocation strategies by region and by crop.

Researchers have developed models to analyze pest control decisions and impacts under a variety of strategies and chemical regulations (see survey by Carlson and Wetzstein, 1993), yet no papers were found examining the efficient allocation of a limited supply of a chemical on a spatial format. In the more traditional regulation case, researchers have found that bans or regulations of particular chemicals may actually benefit producers in some areas while negatively impacting producers in other areas. Analyzing the disaggregated impacts of a pesticide ban by regions and crops, Lichtenberg, Parker and Zilberman (1988) found a large redistribution of income among producers in different regions between pesticide users and nonusers. A pesticide regulation's impact on consumer and producer surplus can be different in the presence of other market distortions, such as price support programs (Lichtenberg & Zilberman, 1986). Other models that look at the effect of pesticide cancellations permit estimation of the welfare changes for competing crop farmers (Taylor, Lacewell & Talpaz, 1979). Liu et al. (1999) suggest that incorporating environmental concerns into locally based Extension herbicide treatment recommendations may be the most efficient mechanism of jointly achieving local farm income and environmental objectives. Alternative regulations such as quotas and marketable permits were found to be welfare-enhancing for the regulation of another fumigant pesticide, methyl bromide (Lynch, 1996; Deepak, Spreen & VanSickle, 1996).

Since the California spatial regulation is not an outright ban (cancellation) or a tax, an allocation strategy is needed to determine who will have the right to use the available pesticide. If the growers' demand for the pesticide exceeds the established limits, chemical manufacturers and distributors might estimate a price in each region to ensure that only the permitted quantity is demanded. Under this method, the chemical companies would earn all the rents of the regulation. A pricing approach may be ineffective unless cross-regional trading can be prohibited. Otherwise, a farmer could simply travel to a nearby region where the chemical is in excess supply and purchase it at a lower price. Alternative allocation strategies include use allocated 1) on a first come, first served basis, 2) on a quota system by crop linked to current or historical use, or 3) on the highest-value use. Use will be allocated on a first come, first served basis in the absence of an alternative distribution scheme, which may result in a suboptimal outcome as higher-valued crops that are planted later in the year and have few available pest control alternatives might effectively be banned from using 1,3-D.

This paper examines the impacts of these three different allocation strategies for the quantity of the fumigant 1,3-D permitted in each geographic area among the variety of crops that use it.[1] California suspended its use in April 1990 after detecting emission levels above air quality standards in Merced County. In 1994, California reintroduced 1,3-D with regulations that set maximum limits on use within spatially delineated areas. These limits currently are binding in only a few areas. Agricultural producers used more than 2.4 million pounds of 1,3-D in California in 1997.

The restrictions on 1,3-D are expected to become binding in more regions as the fumigant methyl bromide is phased out under the Clean Air Act and Montreal Protocol if growers shift to using 1,3-D in its place. In 1992, the Montreal Protocol, an international treaty to reduce ozone-depleting substances worldwide, identified methyl bromide[2] as a class I ozone depletor. Due to this identification, the U.S. Clean Air Act required that all production and importation of methyl bromide cease by January 1, 2001. In 1998, the U.S. Congress amended the Clean Air Act to harmonize the U.S. phaseout with the international schedule under the Montreal Protocol, with reductions of 25% by 1999, 50% by 2001, 70% by 2003 and 100% by 2005.

To assess the economically viable alternatives to methyl bromide for crops that are grown in California using methyl bromide, information was collected through two regional meetings, published research, and numerous interviews with scientists, growers, and policymakers as to the relevant and most likely alternative to methyl bromide. Researchers identified 1,3-D, either alone or in combination with other chemicals, as the best alternative to fumigation with methyl bromide in terms of the lowest per-unit cost (Carpenter, Gianessi & Lynch, 2000). For many crops, herbicides and fungicides will have to be used in conjunction with 1,3-D to achieve the same spectrum of pest control provided by methyl bromide. Although identified as the most likely alternative, the wide-scale availability of 1,3-D as a replacement for methyl bromide is uncertain due to restrictions on its use. As the phaseout of methyl bromide proceeds, allocation of the limited amount of 1,3-D use in each geographic area will be necessary.

The expected demand for 1,3-D is estimated assuming the complete phaseout of methyl bromide. Data from the California Pesticide Use database is used. This database provides detailed spatial information on all pesticide applications as well as application dates. We analyze which crops receive the "right" to use the limited amount of 1,3-D available in each region using the various strategies outlined above. The efficiency and distributional aspects of each strategy based on the factor market impacts, i.e. the value of marginal product and its scarcity or shadow value of 1,3-D, are evaluated. This type of methodology has been used previously to analyze the impacts of the methyl bromide ban (Carpenter, Gianessi & Lynch, 2000; Lynch, 1996; Sunding et al., 1996; Yarkin et al., 1994).

Current restrictions include a limit on the total amount of 1,3-D that may be applied within 36-square-mile areas, referred to as townships (California Department of Pesticide Regulation (California DPR), 1999). These limits and conditions of use were based on a California DPR risk assessment using air quality data on 1,3-D emissions provided by a four-year research and field trial project into modified use practices by DowElanco (California DPR, 1994). The township limit depends on the depth and timing of applications, as these impact emission levels and thus air quality. A total of 90,250 lbs. of 1,3-D per township is allowed if all applications are made to a depth greater than 18 inches between the months of February and November. The permitted quantity is reduced if applications are made at shallow depths or during the months of December or January. The smaller permitted quantities are calculated by counting each pound applied at a shallow depth (i.e. less than 18 inches) during February through November as 1.9 pounds. Similarly, the amount of 1,3-D used is weighted more heavily for applications at any depth during December or January. If all applications are made at a shallow depth between February and November, the permitted quantity is 47,500 lbs.[3]

MODEL

In a first-best world, a directly linked policy instrument would result in growers internalizing the production externality, i.e. the societally imposed costs of air-borne 1,3-D emissions would be part of the production cost calculations and would lead to the optimal level of chemical use from society's perspective. A Pigouvian tax or abatement subsidy might be the optimal instruments to impose on 1,3-D users (Baumol & Oates, 1988). In order to actually apply one of these mechanisms, policymakers would need to measure emissions on each farm and derive the actual costs to society of these emissions in terms of human health and other environmental impacts. The marginal social cost of the pollution and the optimal tax or subsidy cannot be easily determined. Calculating the exact costs related to 1,3-D emissions would be complicated and costly, as would measuring the emissions from each field.

Given the difficulty and expense of calculating the social costs of 1,3-D use, a tax or subsidy mechanism would be untenable. Therefore, the California DPR has chosen to limit use by geographic location as the most efficient way of achieving air quality goals and ensuring emission levels do not exceed the regulated levels. Given a predetermined permitted quantity of use, the distributional and efficiency impacts between regions and crops can be compared under different scenarios for allocating the allowed 1,3-D use among producers in these geographic areas.

Growers in township k choose to grow crop i using the best available pest management technology h. For any individual grower, the market price of the crop i, P_i, the variable input costs, a vector v, and the technology input costs, a vector w, are all exogenous. Growers are assumed to maximize a per acre net revenue function by selecting variable inputs per acre for each crop in each technology, x_{ikh}, and technology inputs per acre for each crop in each technology, m_{ikh}, given production function $f(x_{ikh}, m_{ikh})$ such that $r_{ikh}(P_i, v, w) = max\ P_i f(x_{ikh}, m_{ikh}) - vx_{ikh} - wm_{ikh}$. Thus, net returns per acre, $r_{ikh}(P_i, v, w)$, is a function of crop price, P_i, and input costs, v and w. A township's overall profit, π_k, is maximized by choosing l_{ikh}, the number of acres in crop i in technology h given the net revenue per acre in the different technologies and different crops. The number of acres planted in each township is constrained by the total number of acres in the 36-square-mile township, A_k. Thus implicitly by the action of each grower choosing l_{ikh}, each township solves the following maximization problem:

$$\max\ \pi_k = \sum_i^I \sum_h^H r_{ikh}(P_i, v, w)^* l_{ikh}$$

$$s.t.\quad A_k \geq \sum_i^I \sum_h^H l_{ikh}$$

$$l_{ikh} \geq 0$$

If a particular chemical $h = 1$ is restricted to less than M_k in each location k, each township faces a second constraint,

$$M_k \geq \sum_i^I m_{ik1} l_{ik1}.$$

The Lagrangian in this case with two constraints would be:

$$L = \sum_i^I \sum_h^H r_{ikh}(P_i, v, w)^* l_{ikh} + \lambda_k \left(A_k - \sum_i^I \sum_h^H l_{ikh} \right) + \gamma_k \left(M_k - \sum_i^I m_{ik1} l_{ik1} \right)$$

λ_k is the shadow price for the land constraint (the quasi-rents for land in region k) and γ_k is the shadow price for the technology or chemical constraint. Using the first-order profit maximization conditions, each township would have an optimal land allocation and the optimal use of each technology. The first-order condition for the land variable is:

$$\frac{\delta L}{\delta l_{ikh}} = r_{ikh} - \lambda_k = 0$$

$$\frac{\delta L}{\delta l_{ik1}} = r_{ik1} - \lambda_k - \gamma_k m_{ik1} = 0 \qquad (3)$$

which demonstrates how land will be used for each crop in each technology in a given township based on its net revenue per acre. If the net revenue per acre for a particular crop is less than the quasi-rent for the land in that township, λ_k, no acres of the crop will be grown, $l_{ikh}=0$. If the shadow value of land is positive, then growers in this township would want to plant more acres as they are making a positive net revenue. In addition to the quasi-rent of the land, the shadow value of each additional unit of the regulated chemical for the particular township must be considered when allocating land to growing crops using this technology. As can be seen by the first-order condition for l_{ik1}, the crop must have a sufficiently high net revenue per acre to cover λ_k and γ_k in townships where the chemical restriction is binding to justify its use.

$$\begin{aligned}
&\frac{\delta L}{\delta \lambda_k}=A_k-\sum_{i}^{I}\sum_{h}^{H} l_{ikh}=0\\
&\frac{\delta L}{\delta \gamma_k}=M_k-\sum_{i}^{I} m_{ik1}l_{ik1}=0\\
&\left[\frac{\delta L}{\delta \lambda_k}\right]\lambda_k=0\\
&\left[\frac{\delta L}{\delta \gamma_k}\right]\gamma_k=0
\end{aligned} \tag{4}$$

Using the equations in (4), we find that the shadow value of the land will be positive only if all the land in the township is used to grow crops. If the shadow value is zero, then the number of acres used would be less than A_k, indicating that the net revenue for crops in this township is equal to or less than zero. Similarly, the shadow value of using the chemical will be positive only if demand for the chemical in the township exceeds M_k. If γ_k equals zero, then the constraint is not binding, leading one to believe that the net revenue using the restricted chemical does not exceed the net revenue of an alternative technology.

The choice of technology is incorporated into the problem as a putty-clay model with fixed proportions technology or, as it is often called, a linear response and plateau (LRP) formulation to generate supply by township. This results in the plateau appearance of the supply curve. Each straight portion of the supply curve for a particular crop reflects one township's production of crop i. Berck and Helfand (1990) demonstrate that the linear response and plateau or von Liebig model performs well. The von Liebig functional form assumes that the crop responds linearly to the addition of a limiting input until a different input becomes limiting. Although this function does not allow perfect substitution between inputs, i.e. the crop needs a combination of inputs to grow, it has been shown that a smooth crop production function can be derived from the LRP form by aggregating the effects of heterogenous inputs. Lanzer and Paris (1991) have also demonstrated the validity of the fixed proportions assumption for fertilizers. Green and Sunding (2000) have also used a putty-clay model with survey data in their study on groundwater pumping and seawater intrusion in California's Salinas Valley. While a putty-clay approach does not allow per acre pesticide use to vary – as one would expect it to in years of heavy pest infestation – in the case of both methyl bromide and 1,3-D, growers typically use the label rate specified per acre rather than adjust the pesticide application rate to the perceived pest level. In addition, growers fumigate only once per year before planting the crop. Thus, we expect this type of model to work well for this problem.[4]

Given the LRP formulation, the value of the marginal product equals the value of the average product. In addition to the per unit cost for the chemical, w, growers would be willing to pay the shadow value or quasi-rent of the technology constraint in the township to use the pesticide up to the value of the marginal product. We can use this relationship to determine the highest-value use of the constrained pesticide, i.e. the implicit shadow value of the chemical if the regulation is binding, and thus the most efficient allocation of the chemical between the competing crops in the township.[5]

As regulations change, growers may have to choose alternative pest management technology. Changes in pest management technology will likely alter the per acre yield and costs, and thus net revenues. Using the equations in (3), we find that a grower will be indifferent between using technology 1 and another technology if $r_{ik1} - \lambda_k - \gamma_k m_{ik1} = r_{ik2} - \lambda_k$. Thus without computing the full model, we can calculate γ_k, the shadow value for the constrained pest management technology, using the net revenues from using the restricted pest management input and from the alternative technologies. We compute the maximum shadow value for pest control technology $h = 1$ with respect to technology $h = 2$, the next best alternative as determined by crop and region, that a grower would be willing to pay by crop (Yarkin et al., 1994; Lynch, 1996). Therefore the implicit value of using a unit of the restricted chemical is

$$\gamma_k = \frac{r_{ik1} - r_{ik2}}{m_{ik1}}$$

The change in net revenue is derived from price-weighted changes in per acre yields and changes in per acre costs. The label rate or recommended use per acre for crop i in region k is used to proxy m_{ik1}. For perennial crops, the shadow value is calculated similarly using the change in net revenue over time assuming a discount rate of 4% for the expected length of the crop's life. The net revenue stream for perennials varies by crop; for example, peaches are replanted every 28 years, while almonds are replanted every 40 years.

DATA

The California Pesticide Use database provides detailed spatial information on all pesticide applications, which allows calculation of pesticide use by township. Using 1997 pesticide use data, the current use levels of 1,3-D and methyl bromide in each township are computed. We calculate the expected use of 1,3-D assuming that all major methyl bromide users will switch to 1,3-D (90% of all uses) given the assumptions about application technology and rates outlined in Table 1. Table 2 presents the county-level estimated unrestricted level of demand for 1,3-D as actual pounds, adjusted pounds as applied to the restriction, and number of acres after a methyl bromide ban. Sixteen counties are expected to use 1,3-D, with the major users being Monterey and three San Joaquin Valley counties. If unrestricted, expected demand for 1,3-D use is calculated to increase from 2.4 million pounds to 15.3 million following the

Table 1. Application Rate Assumptions.

Crop	1,3-D Rate (lb./acre)	Application Depth
Almond	332	> 18″
Carrots	97.5	< 18″
Grapes	332	> 18″
Lettuce	76	< 18″
Nursery	235	< 18″
Peach	332	> 18″
Peppers	76	< 18″
Strawberry	235	< 18″
Sweet Potato	190	< 18″
Tomato	76	< 18″
Walnut	332	> 18″
Watermelons	114	< 18″

Source: Kirk Fowler, General Manager, TriCal Inc., Hollister, California.

Table 2. Estimated 1,3-D Demand After Methyl Bromide Phaseout by County.

County	Lbs.	Adjusted lbs.	Acres
Monterey	2,187,757	4,080,302	12,068
Kern	1,675,556	2,733,800	11,064
Merced	1,649,560	2,601,456	7,485
Ventura	1,335,387	2,528,461	6,355
Fresno	1,200,507	1,313,085	4,971
Tulare	917,710	959,242	3,036
Stanislaus	842,319	1,213,464	4,452
Santa Barbara	790,010	1,442,433	3,377
San Joaquin	676,743	861,205	2,586
Santa Cruz	647,303	1,229,876	3,164
Orange	393,905	748,420	1,855
Sonoma	377,538	387,446	1,310
Imperial	338,551	643,246	3,803
San Diego	292,617	555,973	2,448
Sutter	284,061	326,978	983
Riverside	238,309	383,929	1,849
STATE TOTAL	15,262,600	24,047,487	77,404

Table 3. Estimated 1,3-D Demand After Methyl Bromide Phaseout by Crop.

Crop	Lbs.	Adjusted lbs.	Acres
Almonds	1,232,485	1,232,485	4,354
Carrots	1,020,605	1,939,149	10,103
Grapes	1,421,319	1,421,319	5,187
Nurseries	1,572,312	2,872,025	7,111
Peaches	305,467	305,467	1,022
Potatoes	264,134	501,854	1,873
Prunes	189,794	189,794	616
Soil Application	1,849,393	1,979,048	6,154
Strawberries	4,765,296	9,054,062	20,322
Sweet Potatoes	1,109,126	2,107,340	5,994
Tomatoes	192,452	365,659	2,539
Walnuts	336,764	336,764	1,108
STATE TOTAL	15,262,600	24,047,487	77,404

methyl bromide phaseout. Table 3 presents estimates of 1,3-D demand by crop after the methyl bromide phaseout, and includes almonds, grapes, peaches, prunes, potatoes, strawberries, sweet potatoes, nurseries, tomatoes, and carrots. Strawberries are expected to be the major user at 4.8 million lbs., or 9.1 million adjusted lbs.

The 1997 pesticide use data were modified in order to address issues in the reporting of methyl bromide use. First, treated acreage for methyl bromide use in perennial crops may be overstated due to reporting of spot treatments on less than one acre as full acre treatments. Therefore, all records with application rates of less than 50 lbs. per acre were deleted. Second, an adjustment was made for unspecified methyl bromide use. More than 2.5 million pounds of methyl bromide use is not reported as used on any particular crop, accounting for approximately 9,000 treated acres (15% of area treated with methyl bromide). County agricultural commissioners in those counties with a large amount of unspecified use were surveyed for further information on which crops were being fumigated. All unspecified use in Siskiyou County (32,757 lbs.) was assumed to be for strawberry nurseries. Unspecified use in Orange County (250,435 lbs.) was assumed to be strawberry acreage. The breakdown of unspecified uses in Fresno, Madera, and Tulare counties, which together account for more than 1.7 million pounds of the unspecified methyl bromide use, is attributed to perennial crops. Unspecified use in other counties is not included.

When the expected demand for 1,3-D was calculated, fifty-five townships were estimated to be over the limit. Growers would choose to use almost 10 million more adjusted lbs. of 1,3-D than is permitted. While many of the 16 counties have only one township where the restriction binds, others have several. Table 4 outlines the number of townships in each county where the restriction is binding as well as the adjusted pounds of 1,3-D over the limit. Monterey and Santa Cruz counties together have 13 townships where the restriction binds. Merced and Kern counties have 14 townships where the restriction impacts growers. Ventura County has 6.

To compute the per crop shadow values for using the restricted chemical, per acre crop values were obtained from California agricultural commissioner data. Strawberry growers are assumed to switch their pesticide technology from 1,3-D to a combination of metam sodium and chloropicrin, with an increase in cost of $520 per acre (Sunding et al., 1996). Using these alternative chemicals, strawberry growers' yields are assumed to decrease by 5%. Other annual crops' yields decrease 10%, assuming they switch from 1,3-D to metam sodium. Costs for annual crops increase $100 per acre. Perennial quasi-rent values were calculated using yield loss assumptions provided by University of California

Table 4. Counties Where 1,3-D Township Restrictions Are Expected to be Binding.

County	No. of Townships Over Limit	Adjusted lbs. of 1,3-D Over Limit
Del Norte	1	4,760
Fresno	2	67,753
Kern	8	776,586
Merced	6	1,663,506
Monterey	9	2,687,809
Orange	2	271,297
Riverside	1	60,085
San Diego	1	123,162
San Joaquin	3	115,288
Santa Barbara	2	970,649
Santa Cruz	4	743,407
Shasta	1	28,936
Sonoma	1	26,768
Stanislaus	4	148,210
Sutter	1	29,388
Tulare	3	94,405
Ventura	6	1,855,946
STATE TOTAL	55	9,667,953

nematologist Michael V. McKenry. Yield losses for perennial crops were assumed to be between 5 and 20%, depending on whether nematode-resistant rootstocks were used. Yield and cost change assumptions for perennials were also based on Carpenter, Gianessi and Lynch (2000). The quasi-rent value of a pound of 1,3-D varies by crop from a high of $28 for wine grapes to a low of $0.50 for olives. Table 5 contains the computed average values of 1,3-D use by crop relative to the next best alternative.

POLICY SCENARIOS AND RESULTS

The California regulations on 1,3-D can be used as a case study of the impacts of different allocation mechanisms that a state might use to determine which crops receive a limited pesticide within impacted areas. Pesticide use may be allocated in several manners: on a first come, first served basis; through quotas by crop based on current or historical use; or with marketable permits based on highest-value use. The crops in a township that are able to use the limited

Table 5. Average Value of 1,3-D Use by Crop.

Crop	$/lb.	Crop (cont'd.)	$/lb.
Almonds	22.87	Onions	8.01
Apricots	17.00	Oranges	17.00
Beans, Dried	1.89	Ornamental Turf	5.75
Beans, Succulent	1.45	Parsley	7.80
Beets	1.09	Peaches	16.00
Broccoli	4.58	Peppers, Bell	12.12
Brussels Sprouts	6.72	Peppers, Chili	6.69
Cabbage	5.00	Perennial Nurseries	17.85
Cantaloupe	3.16	Plum	17.00
Carrots	4.28	Potatoes	2.71
Cauliflower	5.90	Prunes	5.00
Celery	3.35	Soil Application	12.55
Cherries	17.00	Spinach	3.63
Chinese Radish	5.00	Strawberry	6.97
Cotton	5.00	Strawberry Nurseries	4.66
Cut Flowers	2.77	Sugarbeet	2.15
Grapes	26.38	Sweet Potato	3.51
Grapes, Wine	28.06	Tomatoes	9.35
Lemon	17.00	Tomatoes, Processing	3.86
Lettuce, Head	6.60	Uncultivated Areas	17.00
Lettuce, Leaf	7.06	Walnut	4.50
Melons	3.52	Watermelons	4.63
Nectarine	27.00	Yams	3.50
Olives	0.49		

quantity of 1,3-D will be determined by the allocation strategy implemented. Allocation on a first come, first served basis may result in a suboptimal outcome as higher-valued crops with few available alternatives might effectively be banned from using 1,3-D. Strawberries, for example, are planted in the fall and might not be treated early enough to be assured availability of 1,3-D. A second distribution scenario is to assign each crop within a township a quota based on its use of 1,3-D in proportion to the total demand in the township. For example, if in a particular township 50% of the total demand for 1,3-D is for carrot acreage, then carrots would get half of the quotas. Finally, state officials could distribute the limited amount based on a bidding system for rights. Presumably the users with the highest-value crops or the least cost effective alternative pest control technology would outbid other users. The maximum bid a grower would make is the shadow value per pound of 1,3-D compared to the next best alternative technology. Thus this strategy has the users with the highest-value crops in each township using 1,3-D until the township limit is reached. Growers could potentially trade these rights in the long run if new technologies became available.

Our analysis uses these shadow values for each crop to compare the impact of the alternative distribution mechanisms. Using the computed value of 1,3-D, the aggregate values of the different strategies are compared by county, crop and for the state as a whole. Tables 6 and 7 present the results by allocation mechanisms by county and by crop. Table 8 shows the aggregate results for the state.

One interesting result is that distribution based on the highest-value use does not always maximize the county-level or crop-level aggregate value. As mentioned above, 1 lb. of 1,3-D applied to a shallow depth (less then 18 inches) counts as 1.9 lbs. toward the limitation since shallow applications may result in a higher level of emissions per pound. For shallow-rooted crops, the shadow value γ_k was computed using $m_{ik} * 1.9$ to determine the highest-value use allocation for each pound of 1,3-D actually applied. As a result, we find that in some counties (Del Norte, Orange, Shasta and Sonoma), allocating according

Table 6. Aggregate Value of 1,3-D by County Under Different Distribution Mechanisms.[a]

County	Quotas ($)	First come, first served ($)	High value ($)
Del Norte	988,819	983,514	967,024
Fresno	18,332,894	18,082,382	18,380,115
Kern	10,854,325	10,730,525	11,839,733
Merced	6,094,832	6,321,223	10,279,862
Monterey	11,269,995	10,519,964	11,446,162
Orange	3,557,899	3,656,119	3,630,106
Riverside	3,026,621	3,343,666	3,587,832
San Diego	1,794,254	1,758,477	1,849,911
San Joaquin	7,279,926	6,788,889	7,544,844
Santa Barbara	2,249,173	2,088,394	2,268,681
Santa Cruz	3,016,454	2,868,761	3,175,024
Shasta	1,121,587	1,090,496	1,018,312
Sonoma	17,494,554	17,105,136	16,373,143
Stanislaus	8,462,024	8,076,982	8,514,385
Sutter	2,672,789	2,667,388	2,852,928
Tulare	14,065,707	14,028,563	14,118,287
Ventura	3,843,569	3,175,165	4,028,333

[a] Calculated from shadow values

Table 7. Aggregate Value of 1,3-D by Crop Under Differing Distributional Mechanisms.[a]

Crop	Quotas ($)	First come, first served ($)	High value ($)
Almonds	12,475,773	12,802,625	18,327,554
Broccoli	127,534	140,578	46,155
Carrots	3,092,917	3,097,054	2,885,742
Grapes	44,286,376	44,244,151	43,950,354
Lettuce	796,680	804,052	658,861
Nurseries	11,818,352	11,620,239	11,740,989
Peaches	4,376,826	4,271,840	4,875,922
Peppers	1,534,549	1,855,094	1,867,962
Potatoes	675,189	585,207	628,226
Prunes	917,762	910,624	821,920
Soil Application	29,399,989	29,279,812	29,717,160
Strawberries	18,399,543	15,243,669	19,036,917
Sweet Potato	1,522,511	1,326,709	886,781
Tomatoes	1,413,092	1,579,023	1,214,535
Walnuts	2,137,594	2,106,560	2,162,000
Watermelon	669,550	690,638	596,803

[a] Calculated from shadow values

Table 8. Aggregate Value of 1,3-D Use Under Differing Distributional Mechanisms.

Distribution Method	$
Unrestricted	200,288,999
Quotas	137,995,679
First come, first served	135,155,902
High value	143,744,940

to the highest-value use had a lower aggregate value than a quota-based or first come, first served allocation strategy. Sonoma County has an aggregate value from a quota sytem of $1.1 million more than with the highest-value use allocation strategy. The highest-value strategy has an aggregate value of $0.7 million less than the first come, first served system as well. Sonoma has only one township that binds.

In the other counties (Monterey, Kern and Merced most noticeably), the highest-value allocation achieved the highest aggregate value. The largest difference between the various strategies appears in Merced County, where the aggregate value under a first come, first served system is $4 million less and under a quota system is $4.2 million less than allocation based on highest value. Merced County has 6 townships binding. In Monterey, which has the greatest number of townships binding, the difference between allocation based on highest-value use and on a first come, first served system is $0.9 million. Kern, where 8 townships are binding, had a difference of $1.1 million between highest-value and first come, first served allocation systems, and $1 million between highest-value and quota allocation systems.

Differences among crops under each allocation strategy were also explored to determine distributional issues. Almonds had the largest difference, at $5.8 million between highest-value and quota allocation systems, and $5.5 million between highest value and first come, first served systems. Almond producers could outbid many of the crop producers for the right to use 1,3-D. They also use the chemical late enough in the season that the restricted amount will already have been used. This suggests that the alternative pest control technology for almonds is much less cost effective than using 1,3-D, resulting in a high shadow value. Strawberries would also fare better under a highest-value scenario, at a $3.8 million difference with a first come, first served allocation system, but only $0.6 million compared to a quota system. Strawberry growers also apply 1,3-D later in the year; therefore, under a first come, first served basis, lower-value users may already have used the limited amount by that time. However, even under a highest-value system, strawberries need greater quantities of the chemical, and if all strawberry growers were to switch to 1,3-D, there would not be sufficient quantities for all interested strawberry growers. Under a quota system, some strawberry producers would be restricted from using the chemical unless they could buy the quota rights from lower-value users. Peaches, soil applications, walnuts, and peppers would all be better off if a highest-value approach were used instead of either a first come, first served or a quota system. Nurseries and potatoes would fare better under a highest-value system relative to a first come, first served one, but not relative to a quota system. Sweet potatoes fared worse under a highest-value system than under a quota (− $0.6 million) or first come, first served system (− $0.4 million). Broccoli, carrots, grapes, lettuce, prunes, tomatoes and watermelon also fare worse under a highest-value scenario than under either of the other two scenarios.

The value of having an unrestricted supply of 1,3-D is $200 million, as shown in Table 8. Compared to unrestricted use, using quotas by crop to allocate limited use achieves an aggregate value of $138 million, using first come, first served, a value of $135 million, and using the highest-value use system, a value of $144 million. Therefore if 1,3-D were allocated to be used by the crops with the highest value in each township, the cost of the 1,3-restriction would be $56 million. The first come, first served strategy has the least aggregate value, resulting in a cost of $65 million from the restrictions on use, $9 million more than under the highest-value use system.

Table 9 reports the acres that will be prohibited from using 1,3-D. The state total of prohibited acres is between 26,000 and 27,000 acres regardless of allocation strategy, which is a third of the acreage for which growers are expected to demand 1,3-D.[6] Strawberry growers face the largest loss under restricted use, with more than 13.5 thousand acres (57% of the 1997 California strawberry acreage) forced to use the next best technology under quota allocations and 13.3 thousand acres under highest-value allocation. Under a first come, first served allocation basis, growers of more than 14.6 thousand strawberry acres (62%) must find alternative pest management strategies. Sweet potato growers on 4,605 acres (52% of the 1997 California sweet potato acreage) are restricted from using 1,3-D under the highest-value allocation

Table 9. Acreage Not Allowed to Be Treated with 1,3-D by Crop.

Crop	Quotas	First come, first served	High value
Almond	1,785	1,723	726
Broccoli	317	270	491
Carrots	1,219	1,144	1,918
Grapes	232	219	193
Lettuce	303	291	551
Nurseries	2,113	2,081	2,404
Peaches	104	146	22
Peppers	663	312	511
Potatoes	235	286	191
Prunes	20	25	83
Soil Application	390	410	347
Strawberries	13,541	14,686	13,308
Sweet Potato	3,703	3,986	4,605
Tomatoes	357	140	615
Walnuts	14	37	22
Watermelon	175	125	310
STATE TOTAL	25,867	26,030	27,351

strategy, with almost 4,000 acres (46%) restricted under first come, first served. Growers of about 3,700 acres of sweet potatoes (42%) will be prohibited from using 1,3-D under the quota system. Approximately 2,400 acres of nurseries will be restricted from using 1,3-D under the highest-value allocation, with approximately 2,100 acres restricted under a quota or first come, first served system. Almond and carrot growers will face restrictions on their acreage as well. Under a highest-value allocation method, almond growers would be permitted to use 1,3-D on 1,000 acres more than under either of the other two methods. Carrot growers will be prohibited from using 1,3-D on almost 2,000 acres (0.08%) under the highest-value allocation. They can use it on almost 800 acres more if a first come, first served allocation strategy is followed. Peach growers will be able to use 1,3-D on almost all acreage under a highest-value allocation. Walnut growers fare well under both highest value and quotas.

FURTHER RESEARCH AND CONCLUSIONS

Expected demand for 1,3-D is estimated to increase from 2.4 million to 15.3 million lbs. following the phaseout of methyl bromide. In 55 townships, expected demand is predicted to exceed the permitted level of use under current township restrictions. Growers planting on one-third of the acreage on which 1,3-D would be used were it not restricted will be forced to seek alternative pest management strategies. To date, the California DPR has not established an allocation strategy per se. They have assigned the rights and responsibilities for allocation of 1,3-D use to the pesticide application companies. The rents associated with the chemical restriction could exceed $40 million if 1,3-D is distributed according to its highest-value use. Would it be in society's best interests for private companies to earn these rents if they increase the price to incorporate the scarcity or shadow value? Could the state better use these funds by having growers bid for the right to use the pesticide? If quotas were allocated based on use and resale of quotas were permitted, this could compensate growers for some of the costs imposed by the restrictions.

From a county-level perspective, most counties (13 of 17 where township restrictions are binding) would achieve a higher aggregate value use of 1,3-D by allocating the chemical to the highest-value use. Three counties fare better with allocation based on proportional quotas. Examining individual crops, 6 would fare better under a highest-value system, while 5 would fare better under a quota system.

These gains to growers by crop or by county assume no transaction costs. The gains to the state, counties, and growers may dissipate through the costs of establishing and implementing an allocation mechanism. Considering the modest difference in aggregate impacts between the three allocation schemes reported in Table 8, the mechanism with the lowest administrative cost may be preferable. Allocation based either on quotas or on a first come, first served basis may require lower transaction costs than a bidding system. In addition to transaction costs under these allocation schemes, enforcement and monitoring costs may be quite large. Allowing the pesticide companies to determine when the limit on use has been reached may be the least costly to monitor and enforce.

The analysis is limited by the assumption that no crop will be geographically relocated following the methyl bromide phaseout and the subsequent restrictions on 1,3-D use. Yet some crops may be able to be shifted to another township where the township restriction is not binding. Growers however are assumed to already be growing the most profitable crop in the most profitable area considering both soil conditions and climate. Therefore, any shift would likely decrease profits. In some cases, crops may not be able to be relocated in the short run because the growers may own the land they farm, restrictions in the nearby township may also be binding, or there may be no other region characterized by the optimal growing conditions. There is also some indication that growers are shifting the application technique due to the township limits. For example, all carrot growers are now applying 1,3-D at a depth greater than 18 inches in order to treat more acres than they would be able to using shallow application, which is preferred. New application methods that decrease 1,3-D emissions may allow California to increase the permitted use. For example, application of 1,3-D through drip irrigation systems may reduce emissions from treated fields and thus may increase the amount of 1,3-D permitted without impacting air quality. An emulsified version of Telone C-35 (1,3-D plus chloropicrin) has received a full federal registration and research on using drip irrigation systems continues.

This analysis included only 90% of areas currently treated with methyl bromide. To the extent that any of the remaining acres switch to 1,3-D after the phaseout, this analysis may underestimate the impact of the township restrictions. In addition, the analysis does not permit price adjustment as yields decrease. Less than 10% of acreage for most crops in 1997 would have been prohibited from using 1,3-D under any of the allocation strategies. Yield losses on these acres would have been assumed to be between 5 and 10%. Therefore including price changes would probably not alter the conclusions significantly. However, for strawberries and sweet potatoes, the affected acreage is a much larger percentage. Both of these crops will see over 40% of their acreage impacted by the restrictions; thus at a minimum, the overall crop supply will decrease 2%. Among the crops themselves, there would not be much difference in aggregate values between the different allocation strategies even if the price changes were incorporated. However, strawberries and sweet potatoes may fare relatively better in the highest-value allocation strategy if their price changes are included.

Since these regulations are being imposed due to air quality concerns and impacts on human health, a modification of the regulation may more closely link the benefits and costs of the pesticide regulation. For example, the regulation might impose tighter restrictions in areas where the external costs of the pesticide use are highest, for instance in more highly populated areas, and allow higher-use levels in less populated areas. A system using marketable permits or taxes to divert use from more populated areas to less populated areas might be explored. Each township is entitled to its limited amount, which growers can use or sell to another area if the "receiving" area is less populated. This mechanism may permit higher levels of use in more rural areas, which could have high-value crops such as strawberry nurseries.

NOTES

1. 1,3-D acts primarily as a nematocide but also controls viruses, bacteria, soil insects and fungi.
2. Methyl bromide is an agricultural fumigant that is widely used in California to control nematodes, fungi, other pathogens, insects, and weeds.
3. In addition to the township caps, growers must maintain a 300-foot buffer zone around occupied structures, which may be greater than the current buffer zone requirements for methyl bromide treated acres. Growers must also meet soil moisture requirements that may reduce the efficacy of 1,3-D. The maximum application rate permitted for 1,3-D is 24 gallons per acre for tarped fumigation and 35 gallons for untarped fumigation.
4. Methyl bromide demand per year has varied between 1996 and 1999. In 1996, use was over 16 million pounds. Then in 1998 use decreased to under 15 million pounds, then climbed again to 16 million pounds (Trout, 2001). Calculations of the relative efficiency of the distribution mechanisms may be affected by which crops have decreased and then increased the use of methyl bromide.
5. This approach assumes constant prices which from an industry perspective may not be realistic if yield and acreage changes are large. Growers may find that the relative profitability of a crop changes and switch to another crop. Depending on price elasticities, price changes may compensate for some of the decreased yield and/or increased costs following the technology change.
6. Total irrigated vegetable acres in California was 1.2 million in 1997 (USDA NASS, 1997) and fruit and nut trees planted with fumigation in any one year is 10,783 acres (Carpenter, Gianessi & Lynch, 2000), for a potential of 1.21 million acres impacted.

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11. PESTICIDE AVOIDANCE: RESULTS FROM A SRI LANKAN STUDY WITH HEALTH POLICY IMPLICATIONS

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ABSTRACT

In this chapter the contingent valuation method is used to estimate the yearly value to an average farmer in Sri Lanka of avoiding direct exposure to pesticides and the resulting illnesses. The costs are shown to be high. The pesticide cost scenarios calculated from the contingent valuation bids for the entire country show that the costs run into millions of Sri Lankan rupees each year. The last section of the paper identifies the factors that influence the willingness to pay (WTP) to avoid direct exposure to pesticides and the resulting illnesses. The health policy implications stemming from the regression analysis are also discussed.

1. INTRODUCTION

Farmers handling and spraying pesticides using hand sprayers suffer from numerous morbidity effects (Jeyaratnam, 1990; Antle & Pingali, 1994; Pingali & Roger, 1995; Sivayoganathan et al., 1995; Owen et al., 1997; Crissman et al., 1998). Recent estimates quoted by the Food and Agriculture Organisation (2000) from Pesticide Action Week (PAN) show that approximately three

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million people are poisoned and 200,000 die from pesticide poisoning each year. The largest number of deaths are in developing countries. As Forget (1991), the World Resources Institute (1998, 1994) and Crissman et al. (1998) point out pesticide poisoning is a major occupational hazard for farmers and their families. The environmental costs of pesticide use are also high (WRI, 1998, 1994; Mullen et al., 1997; Crissman et al., 1998; Cuyno, 1999; Wilson, 2000). In addition, pests have developed resistance to pesticides (Widawsky, 1998; Wilson & Tisdell, forthcoming). The direct, indirect and intangible costs arising from exposure to pesticides alone incur substantial costs to farmers. In this chapter the contingent valuation method (CVM) is used to determine the yearly value to an average farmer in Sri Lanka of avoiding direct exposure to pesticides and the resulting illnesses. In other words, the CVM is used to obtain willingness to pay (WTP) bids for a year of avoiding the costs arising from direct exposure to pesticides and the resulting illnesses. CVM is employed because this method captures both the tangible as well as the intangible costs such as discomfort, stress, pain and suffering. These are important costs that are associated with exposure to pesticides. From the CVM bids obtained, the cost scenarios for farmers in Sri Lanka can be calculated to show the magnitude and severity of pesticide poisoning. Although it is important to use toxicological information to develop interventions, it is also important to show the economic costs of such poisoning. As Higley and Wintersteen (1992, p. 34) point out in relation to environmental risks from pesticide use "Estimating the economic costs of environmental risks is essential for weighting differences between risks and for integrating environmental and economic data". They go on to state that only contingent valuation seems applicable in measuring all the environmental costs associated with the use of pesticides. From the field data collected it is also possible to identify the factors that influence the WTP to avoid direct exposure to pesticides and the resulting illnesses among subsistence farmers.

2. PESTICIDE USE AND HEALTH PROBLEMS IN SRI LANKA

Since the first use of pesticides in Sri Lanka in agriculture in the 1950s and their increasing use after the introduction of high yielding varieties (the so called Green Revolution technology) and commercially grown cash crops (e.g. vegetable crops), the health of farmers, too, have been greatly affected from exposure to pesticides during handling and spraying on the farms. Deaths are also not uncommon (Fernando, 1991; Wilson, 1998). Hospital statistics show that on average 14,500 individuals were admitted to government hospitals and around 1,500 individuals a year died from pesticide poisoning in Sri Lanka during the period 1986–1996 (National Poisons Centre, 1997). However, not all hospital admissions and deaths were due to occupational poisoning (i.e. due to handling and spraying on the farms) but include cases of self ingestion (suicides), accidental ingestion and homicides as well.[1] Apart from these hospital data, various field studies conducted, too, have confirmed the high levels of morbidity from direct exposure to pesticides ranging from feeling faint, headaches, nausea, diarrhoea, muscle twitching, rashes and cramps (Jeyaratnam et al., 1987; Rola & Pingali, 1993; Crissman et al., 1994; Dharmawardena, 1994; Pingali et al., 1994; Sivayoganathan et al., 1995; Crissman et al., 1998). Antle et al. (1998), Crissman et al. (1998) and Pingali et al. (1994) are studies that deal with the effects of pesticides on health using field survey medical data. These are the short-term symptoms recorded during or soon after spraying pesticides. There are many short-term illnesses that arise on non-spraying days as well. Numerous studies in the United States have also documented long-term illnesses arising from exposure to pesticides (e.g. Hoar, 1986; Neilson & Lee, 1987; Blair & Zahm, 1993). The severity of short-term illnesses experienced by farmers on spraying and non-spraying days can be grouped into three categories, namely severe, moderate and mild.[2] In all of these categories, respondents suffer private[3] direct, indirect and intangible costs. The direct and indirect costs can be further subdivided into medical costs which include doctor visits, hospitalization and laboratory costs, emergency room visits and medication/drug costs. These are categorized as direct costs. Other direct costs include dietary expenses resulting from illnesses, travel costs associated with medical treatment, hired labour due to inability to work on the farm, inability to look after the crops from animal damage and theft. The indirect costs are loss of work days on farm, loss of efficiency on farm, time spent travelling/seeking treatment and leisure time losses. The intangible costs include pain, discomfort, stress and suffering.

The field study which was carried out in the summer of 1996 revealed that 96% of the respondents had suffered some form of after-effect during the past year from pesticide poisoning (excluding effects on non-spraying days or long-term effects). However, such after effects did not necessarily lead to hospitalization or taking treatment from a physician. Nevertheless, costs were incurred such as those due to self-treatment, loss of working days, loss of efficiency at work and loss of leisure time. Table 1 shows the extent of the costs arising from direct exposure to pesticides and the costs of precautionary measures taken. The costs of different categories of ill-health experienced by a farmer are not mutually exclusive. In other words, a farmer who is hospitalized in a given year can, after returning to work fall sick again in the same year from exposure to pesticides and hence be re-admitted to hospital, take treatment

Table 1. Number of Respondents Incurring Costs due to Exposure to Pesticides in the Study Area.

	Beligamuwa		**Ambana**		**Kandalama**		**Yatawatte**		**Polonnaruwa**		**Total**	
Respondents	42		31		46		53		31		203	
	No	**%**	**No**	**%**	**No**	**%**	**No**	**%**	**No**	**%**	**No**	**%**
Medical and other Costs												
A	13	30%	06	19%	08	17%	08	15%	06	19%	41	20%
B	09	21%	04	13%	23	50%	22	41%	4	13%	62	30%
C	33	78%	30	97%	20	43%	25	47%	28	90%	136	64%
NSD	21	50%	14	45%	34	73%	14	26%	04	13%	87	42%
LTC	09	21%	07	22%	23	50%	25	47%	07	23%	71	35%
Defensive Costs												
PC	20	48%	31	97%	32	69%	25	47%	16	51%	123	61%
OC	04	10%	09	29%	21	46%	26	49%	03	10%	66	32%
All	22	52%	31	100%	32	69%	40	75%	17	55%	142	70%
EP	42	100%	31	100%	46	100%	49	92%	27	87%	195	96%

A: Respondents admitted to hospital and incurring private costs (includes all costs associated with pesticide pollution).
B: Respondents consulting a doctor and incurring private costs (includes all costs associated with pesticide pollution).
C: Respondents not admitted to hospital or consulting a doctor, but seeking some form of treatment and incurring private costs (includes all costs associated with pesticide pollution).
NSD: All private costs incurred on non-spraying days due to exposure to pesticides (includes costs of medicine, consultation and other costs).
LTC: All long-term private costs incurred due to direct exposure to pesticides (includes costs of medicine, consultation and other costs).
PC: Number of respondents incurring costs of some form of protective gear.
OC: Number of respondents incurring costs apart from costs of protective gear (for example, costs incurred on special storage and hiring labour).
ALL: Includes all respondents incurring costs on protective clothing and other defensive behaviour.
EP: Number of respondents suffering from acute illnesses described in the interview on a typical pesticide spraying day (excludes non-spaying days and long-term illnesses) and incurring costs. There were eight respondents in the sample (n = 203) who did not incur any costs.
Note: The costs of different categories of ill-health experienced by a farmer are not mutually exclusive.

from a doctor or home-made treatment and incur costs. In fact, the survey revealed that a farmer who has suffered a serious illness (hospitalized) from exposure to pesticides was more likely to fall sick when she or he returned to work and sprayed pesticides once again on the farm. The table shows that during 1996, 20% of the farmers interviewed had been admitted to hospital and incurred costs. These were farmers who suffered ill-health during or soon after spraying (usually within four hours). Another 30% had taken treatment from a doctor and incurred costs while 64%, incurred private costs due to home made self treatment. Furthermore, 42% of the respondents incurred illness-related costs on non-spraying days and 35% incurred costs due to long-term illnesses resulting from direct exposure to pollution. It must be pointed out here that these are costs that were incurred during the whole of 1996. They are not costs incurred during a single spraying day.

As Table 1 shows, farmers also incur precautionary/defensive costs. When all these costs are aggregated, they are substantial. Therefore, it is necessary to estimate these costs to show how large and significant they are to farmers. Many techniques have been suggested that can be used to estimate these costs. Three of the commonly used techniques are the cost of illness, avertive behaviour and the contingent valuation approaches. The former two approaches have several limitations including their inability to take into consideration intangible costs such as discomfort, stress, pain and suffering (see Wilson, 1998 for a discussion of the limitations of these two approaches). However, the latter approach can take into consideration the intangible costs such as pain, suffering, discomfort and stress associated with an illness. These are important costs that need to be taken into account since those suffering from exposure to pesticides undergo considerable pain, stress, suffering and discomfort. These effects are also known to result in suicides among farmers exposed to pesticides. Furthermore, it is now known that women and children in the families of those who spray pesticides are being adversely affected through indirect means such as pesticide-covered clothes, poor living conditions, poor health and diet (WRI, 1998). In this work, the CVM is used to ask farmers the value of avoiding direct exposure to pesticides and the resulting illnesses for a year. In other words the value of avoiding the costs of direct exposure to pesticides and the resulting illnesses are estimated. The values expressed by the respondents represent the costs the individual thinks he would incur from exposure to pesticides. An individual who has suffered from exposure to pesticides would consider all the costs arising from ill-health including the resulting pain, stress, discomfort and suffering and all the costs of defensive action taken to minimize exposure to pesticides. The next section briefly discusses the CV approach. The main strength of this technique in the field of health economics is to capture intangible and invisible costs such as pain, discomfort, stress and suffering despite the drawbacks that have been discussed in the literature on CVM.

3. THE CONTINGENT VALUATION APPROACH AND THE VALUATION OF MORBIDITY EFFECTS

Since the first application of the contingent valuation technique by Davis (1963),[4] it has been widely used during the last few decades to estimate economic values for a wide range of commodities for which there is no market. In the last decade, there has been a dramatic increase in the number of academic papers and presentations using the contingent valuation technique including many studies conducted in developing countries.[5] Whittington (1998) points out that the CVM can be applied to obtain values of pure public goods, goods with both private and public characteristics and private goods. Contingent valuation in the 1990s was a well established and widely employed technique for valuing non-market goods and is supplemented by other techniques of measuring values of non-market goods.

The appeal of the contingent valuation method is that, in principle, it can elicit WTP bids from a broad segment of the population, and can value causes of deaths and illnesses that are specific to environmental hazards or a specific disease category. This method has been recommended especially for the estimation of values and costs that are difficult to estimate such as non-use values (e.g. bequest and existence values) and intangible costs (pain, discomfort, stress and suffering) where there are no direct market transactions taking place which can be used to estimate economic values. This technique tries to cover such a void. In this chapter, for example, the farmers were asked what they would be willing to pay for a year to keep them free of health risks arising from direct exposure to pesticides during handling and spraying. In other words the value of avoiding the costs of pesticide related illnesses.

Although initially the CVM was developed to measure the value of non-market goods such as the value of recreation and the environment, it has been adopted by economists to measure the value of risk reductions, too, and in recent years, an increasing number of studies have been conducted by health economists to assess the value of health care and the cost of illnesses (e.g. Donaldson, 1990; Johannesson et al., 1991; Johannesson, 1992; Johannesson et al., 1993; Kartman et al., 1996; Sloan et al., 1998; Zethraeus, 1998).[6]

Many contingent valuation studies have also been conducted to determine the value of avoiding symptoms associated with environmental pollution. Some studies carried out to value morbidity effects (such as headaches, eye irritation, sinus congestion, wheezing and nausea), both minor and acute, associated with air pollution include Loehman et al. (1979); Rowe and Chestnut (1985); Tolley et al. (1986); Dickie et al. (1987); Chestnut et al. (1988); Chestnut et al. (1996); Alberini et al. (1997) and Alberini and Krupnick (2000). CVM that have been conducted to determine the value of avoiding or reducing pesticide related health (mostly relating to pesticide residues in food) and environmental risks include: van Ravenswaay (1991a); van Ravenswaay (1991b); Misra et al. (1991); Higley and Wintersteen (1992); Hammitt (1993); Buzby et al. (1995); van Ravenswaay and Wohl (1995); Mullen et al. (1997); Cuyono (1999); Fu et al. (1999). An example of a contingent ranking method (another stated preference technique) that has been used to value multiple impacts of pesticide use is found in Foster and Mourato (2000).

The contingent valuation survey technique, because of its ability to consider non-use values and intangible costs is widely used for the estimation of environmental and health benefits. In the next section, the manner in which the CVM was administered to obtain the contingent valuation bids to avoid direct exposure to pesticides is discussed. From the bids elicited, the private costs to an average farmer for a year arising from direct exposure to pesticides and the resulting illnesses are estimated.

4. CONTINGENT VALUATION BIDS TO AVOID DIRECT EXPOSURE TO PESTICIDES

A field questionnaire was carefully designed to gather data on direct exposure to pesticides and a section of it was devoted to obtaining contingent valuation bids to avoid the costs of direct exposure to pesticides and the resulting illnesses. The standard procedures recommended by Mitchell and Carson (1989); NOAA (1993) were followed. The respondents were presented with well defined situations of exposure to pesticides which they encounter on a given pesticide spraying day. The respondents were provided with details of pesticide exposure situations which the respondent was asked to value as recommended by Mitchell and Carson (1989). The interviewer explained in detail the health hazards faced by small-scale subsistence farmers handling and spraying pesticides on a regular basis. Reference was made to the fact that current high levels of direct exposure to pesticides have a high probability of causing many side effects and even deaths. It was explained to the farmers that the risks of ill-health increase with the levels of direct exposure, i.e. due to longer hours of spraying, larger acreage sprayed, potency of the pesticides used and the level of precautions taken and so on. Prior to asking the contingent valuation question, data on costs of illnesses and defensive behaviour were gathered. Furthermore, information on the health status of the interviewee was also obtained using the same list of illnesses used in the Sivayoganathan et al. (1995) study. In the Sivayoganathan et al. (1995) study, a physician examined the commonly occurring short-term illnesses during spraying or soon after spraying. In this study, farmers were asked how often they suffered from any of the 17 identified symptoms in the Sivayoganathan et al. (1995) study, or any other symptoms on an average pesticide handling and spraying day. Information was also obtained on illnesses that arise on non-spraying days and long-term illnesses arising from exposure to pesticides. Only illnesses diagnosed by physicians as arising from direct exposure to pesticides or those illnesses which farmers can strongly attribute to the use of pesticides have been considered.

The respondents were told that the CVM question was aimed at measuring how much people are willing to pay to avoid direct exposure to pesticides and the resulting illnesses if a programme was devised to prevent such illnesses from direct exposure. Respondents were also informed of the economic sacrifice they would have to make to support such a prevention programme. The farmers were told that the money will have to come out of their income or from some other income source. They were specifically told about the range of options available to avoid direct exposure to pesticides (for example, using safer but more expensive pesticides, adopting integrated pest management (IPM) strategies which, however, could cost more to adopt, hiring labour to spray pesticides and growing crops that involve no or less use of pesticides). The choice of the payment vehicle to undertake prevention programmes was also made as realistic as possible. Taxes were deliberately avoided because during the pre-testing of the questionnaire (pilot study), it was found that respondents disliked the idea of taxes[7] and thought that this study was being conducted to compile a register for the implementation of taxes in the future. Therefore, due to such difficulties, higher prices/costs were preferred to taxes.[8] Interestingly, Carson [per com. (1998)] points out that a major problem with contingent valuation surveys in developing countries is that of finding a plausible payment vehicle for the good in question. All the respondents in the study areas were provided with the same information, including the payment vehicles suggested. An open-ended question format asking what the maximum amount farmers were willing to pay in order to avoid direct exposure to pesticides and the resulting morbidity effects was used.[9] The questionnaire was translated into Sinhalese (the main Sri Lankan language) and the data was obtained by direct interviews. The CVM question was framed as follows:

Question: In view of the large short-term and long-term and precautionary costs which we saw in the preceding sections of the questionnaire, what is the yearly value to you of avoiding direct exposure to pesticides and the resulting illnesses. In other words what would you be willing to pay for a given year to avoid costs arising from ill-health.

Please state the highest money value in Rupees . . . .

The basic objective of the sample design of the questionnaire was to obtain data from a representative cross section of farmers to base inferences about pesticide use and the numerous health effects arising from such use and their costs. This also involved obtaining contingent valuation bids. The period from June 1995 to June 1996 was considered. Five areas were sampled from the intermediate dry zones of Sri Lanka where intensive agriculture is widespread. The regions covered were Yatawatte, Kandalama, Beligamuwa, Ambana and Polonaruwa in the Central and North Central provinces of Sri Lanka, within a 75–100 mile radius. Only farmers who are regular pesticide users and cultivate land not less than half an acre and not more than three acres were selected. This is because according to a census carried out in 1982 by the Department of Census and Statistics, the average size of land cultivated in the country was 1.94 acres. Therefore, as the census statistics show, a large number of farmers cultivate a land area which is less than three acres and more then half an acre. The five regions selected specialize in growing food crops. As a result, the level and intensity of pesticides used and the level of direct exposure to pesticides vary from region to region. Prior to the interviews a pilot study was conducted in the study area. A total of 27 such interviews were conducted, which made it possible to check out the viability of the questions prepared to collect the necessary data. As a result, the questionnaire had to be modified greatly, removing questions that proved difficult to administer.

Judgment sampling (which is a non-probability sampling technique) was employed to collect the data necessary for the study. This was owing to the impossibility of carrying out a simple random sampling study for the entire country due to financial and time constraints. Instead, judgment sampling was resorted to, according to the information and advice given by officials of the Department of Agriculture. Another reason that influenced judgment sampling was that the agriculture officials who were contacted were of the view that the regions they recommended were the best for the planned study, as they were representative of the farming community. These regions were representative of the farming community because pesticides were used on a regular basis by farmers and they were affected by direct exposure. Furthermore, a previous study (Sivayoganathan et al., 1995) had been carried out almost in the same areas covered by this study. The Sivayoganathan et al. (1995) study revealed health problems resulting from direct exposure to pesticides. The five areas from the intermediate and dry zones, covering two provinces, represented a large and diverse group of farmers growing a wide variety of crops (hence the quantity and variety of pesticides used). The sample chosen, therefore, was a cross section of farmers using pesticides in the country.[10] Although lists of farmers in the study area were available, they were found to be unreliable in selecting farmers who use pesticides on a regular basis. It was found that either the farmers were absent on the land, cultivated a small plot of land, did not use pesticides on a regular basis or were not present on the farm the days they were visited. Hence, it was not possible to resort to simple random sampling techniques in collecting the data. Once again, non-probability sampling procedures had to be adopted. Convenience sampling, which was the best option available given the problems mentioned earlier, was resorted to in order to obtain the required samples. Hence, the interviewer selected the samples from the areas under study. This was done as follows: after visiting a village in the study area, the interviewer walked into a farm randomly and the farmer was asked whether they cultivated a land area which was not less than half and not more than three acres. If they did, then they would be interviewed. Otherwise, another farm was chosen. Once an interview was completed (which was the only one conducted for the whole morning), another farm was selected from the same village (usually within one or two miles) for the afternoon, which too, fulfilled the requirements of the field study specified earlier. Very rarely were the interviewers able to conduct three interviews on a given day. On each day, a different part of the village was chosen. Once a village was sufficiently covered, another village was visited and the same procedure was applied. During the entire study period, a large number of the villages in the study areas were covered.

Initially, 227 farmers were interviewed, out of whom, one respondent refused to give a bid and two gave zero bids. One of the respondents who gave a zero bid was found to be the father of the owner of the pesticide shop in the village. It was believed that he had an interest in the son's business. This was because he suffered from mild symptoms from exposure to pesticide spraying, but yet gave a zero bid to avoid direct exposure to pesticides. There was another zero bid. Although this respondent, too, had suffered from ill-health due to direct exposure to pesticides. Because of lack of consistency of these two bidders, they were removed from the sample. The protest bid was also removed from the sample as recommended (Hanley & Spash, 1994). Twenty one questionnaires had not recorded either household incomes, age, education, household size and acres sprayed which are important variables used in the

Table 2. Contingent Valuation Bids for the Study Areas.

Sample Group	Sample Size	Lowest Bid Rs	Highest Bid Rs	Average Bid Rs
Study Sample	203	300	70,000	11,471.18
Ambana	31	300	38,000	12,829.03
Kandalama	46	500	50,000	12,834.78
Polonnaruwa	31	1000	65,000	15,370.97
Yatawatte	53	300	70,000	7,548.11
Beligamuwa	42	600	50,000	11,047.62

regression analysis in the last section of this paper. They, too, were removed from the sample. For the entire study, the contingent valuation bids varied a great deal from bids as low as Rs 300 to as high as Rs 70,000 (see Table 2). The amounts bid varied across individuals according to the extent of direct exposure to pesticides and the severity of the illness suffered, income earned, and a host of other factors. On average, farmers who were often exposed to pesticides and who suffered a great deal made larger bids, while those with less exposure and who suffered fewer health effects, bid less. Furthermore, farmers with higher incomes made higher bids. Once the necessary bids were obtained, the average WTP was calculated for the sample under study. The average contingent valuation bid for the sample group was Rs 11,471.18.[11] The payment was not in the form of a one-off payment per year but was, for example, in the form of higher prices paid for safer pesticides which farmers would purchase 5–8 times a year.[12] When the farmers were asked why they were willing to pay this figure, the reason given was that there is extreme suffering and costs associated with direct exposure to pesticides. In the next section it is briefly explained why farmers continue to use pesticides despite the high health and environmental costs. This issue is discussed in detail in Wilson and Tisdell (forthcoming).

5. WHAT ACCOUNTS FOR THE FAILURE TO SHIFT AWAY FROM CHEMICAL PESTICIDES?

As shown in the previous section, the costs arising from handling and spraying pesticides are high to farmers. In such a case, the question that is, often asked is why do farmers continue to remain in agriculture and use pesticides? One reason that can be given is that farmers in developing countries have no easy alternatives to subsistence farming. Subsistence farming on the other hand requires very little capital and skill. Furthermore, another advantage is that subsistence farmers use some of their produce for home consumption thus covering a large part of the family expenditure. It is also likely that in the majority of cases, the short-term health effects arising from pesticide use and the disutility from that ill-health are underestimated by farmers since the costs mentioned in Section 2 accrue over a period of time (e.g. one year) and include time costs as well. It is also likely that advertising and promotion by pesticide companies create a bias in favour of their use (Tisdell et al., 1984). Another possible reason that can be cited is that as labour costs increase, farmers are driven to substitute herbicides for hand-weeding in rice, vegetables and other crops. Furthermore, although farmers in this study were willing to pay a higher price to use safer pesticides or adopt Integrated Pest Management (IPM) strategies such services are not easily available to farmers in developing countries. IPM is practiced in developing countries but has been on a small-scale due to many reasons.[13] As the WRI (1994) points out, IPM in developing countries is more the exception than the rule. Tisdell (1991) points out that when chemical agricultural systems are adopted, agricultural yields or returns become very dependent on them despite the very high costs and thus impose an 'economic barrier' to switching to organic systems. In short, agricultural practices tend to become 'locked' into such systems once they are adopted despite being unsustainable (Tisdell, 1991; Tisdell, 1993; Wilson & Tisdell, forthcoming). Cowan and Gunby (1996), too, point out that once a pest control strategy is adopted, it becomes the dominant strategy as has been the case with using chemical pesticides. They point out that once the chemical pest control strategy was adopted, the amount of money spent on R&D for further development of pesticides has increased while the development of IPM has slowed down. For example, they show that "in 1937, 33% of the articles in the Journal of Economic Entomology dealt with the general biology of insects, 58% were devoted to testing pesticides. By 1947 these proportions were 17% and 76% respectively" (p. 524). As a result, in a competition between two technologies, "a lead in market share will push a technology quickly along its learning curve, thereby making it more attractive to future adopters than its competitor. A snow-balling effect can lock a market of sequential adopters into one of the competitors" (p. 523). The use of chemicals can also affect IPM strategies by killing the predators of pests. Hence, even if some farmers decide to adopt IPM strategies, they would be affected due to externalities of pesticides arising from neighbouring farms. Therefore, despite the economic, social and ecological gains that could be derived from IPM (Menz et al., 1984; Tisdell, 1987, 1990), pesticides once adopted as the dominant pest control strategy will continue to be used in larger quantities despite the very serious negative effects that have arisen.[14] For example, Cowan and Gunby (1996) state that between 1964 and 1982 in the United States, the application of active chemicals increased 170% by weight. Since 1970, herbicide use has more than doubled. In Sri Lanka pesticide use has increased by almost 110 times between 1970 and 1995 (Wilson, 1998). In addition to the increase in quantity of pesticides used, farmers use stronger concentrations of pesticides, they have increased the frequency of pesticide applications and also mix several pesticides together to combat pesticide resistance by pests (Chandrasekera et al., 1985). Therefore, despite the advantages of IPM strategies, most farmers both in developed and developing countries continue to use pesticides at an increasing rate and hence become 'locked in' on one form of pest control technology which has resulted in their 'entrenchment' in pesticides. However, it is worth mentioning that some countries such as Indonesia, Guatemala, Sweden, Norway, Denmark and the Netherlands have in fact reduced the use of pesticides without diminishing crop yields (Pimentel, 1997). The economic costs, however, have been large. For instance, Indonesia in the late 1980s invested as much as $1 million a year in IPM/ecological/biological research, followed by extension programmes to train farmers to conserve natural predators of pests.

6. CONTINGENT VALUATION COST SCENARIOS FOR THE ENTIRE COUNTRY

It is now possible to use the WTP bids shown in Table 2 to estimate the contingent valuation cost scenarios for the entire country. It is necessary to resort to scenarios because no government agency in Sri Lanka, including the Department of Agriculture and the Department of Health or the Pesticide Poisons Centre, know the number of farmers affected by direct exposure to pesticides during handling and spraying on the farms. This is a common problem in many developing countries. For the scenarios in this study, we use 1978 employment survey data compiled by the Department of Labour (1978) which put the number of agricultural workers in Sri Lanka at 472,435. A census carried out in 1982 (Department of Census and Statistics, 1983) estimates the number of 'agricultural operators' at 1,803,99. An agricultural operator has been defined as any person responsible for operating an agricultural land or one who looks after livestock or poultry. The agricultural land defined includes all plantation crops such as tea, rubber and coconut and cash crops as well where pesticide use is minimal. This also includes home gardens and land not cultivated on a regular basis. The owner of any of these lands, or a person engaged in livestock or poultry farming, is also classified as an 'agricultural operator'. Since this definition of agricultural operators is wide, the use of employment survey data of 1978 is used.[15] Since these two surveys were conducted, no survey has been conducted to determine the number of agricultural workers in the country. This is due to the continuing civil war in the North-East of the country which started in 1983. Of the 472,435 agricultural workers in Sri Lanka (according to the 1978 employment survey), not all use pesticides since some of them are plantation workers. It is assumed that a minimum of 50,000 and a maximum of 300,000 agricultural workers are affected each year due to direct exposure to pesticides in Sri Lanka. Table 3 shows such cost scenarios for the entire country.

The lowest contingent valuation bid estimates show that the value to farmers in Sri Lanka of avoiding direct exposure to pesticides, or in other words the cost of direct exposure to pesticides, is more than Rs 573 million (scenario A) while the high value/cost scenario (scenario D) indicates that farmers incur a cost of more than Rs 3,441 million in the form of costs due to direct exposure to pesticides. These costs include not only the direct and indirect costs of direct exposure to pesticides, but includes intangible costs as well.

The contingent valuation approach used in this study conformed to all but one of the appropriate and applicable guidelines laid down by the NOAA panel for such studies, including the main guidelines as identified by Portney (1994). The contingent valuation study, however, could not adopt a referendum format, the reasons for which were given in footnote (9) earlier in this section.

Table 3. Contingent Valuation Cost Scenarios for Sri Lanka (in Millions of Rupees).

Sample Group	A Rs	B Rs	C Rs	D Rs
Study Sample	573.559	1147.118	1720.677	3441.354
Ambana	641.451	1282.903	1924.354	3848.709
Kandalama	641.739	1283.478	1925.217	3850.434
Polonnaruwa	768.548	1537.097	2305.645	4611.291
Yatwatte	377.405	754.811	1132.216	2264.433
Beligamuwa	552.381	1104.762	1657.143	3314.286

Note: The average contingent valuation bids are multiplied by the number of farmers whom we believe are affected by direct exposure to pesticides. Harrington et al. (1989) study, too, adopt a similar approach to estimate costs in their study. We believe between 50,000 to 300,000 farmers are affected. Accordingly, we prepare the scenarios as follows: Scenario A = 50,000 farmers. Scenario B = 100,000 farmers. Scenario C = 150,000 farmers. Scenario D = 300,000 farmers.

There are several ways through which the validity of the contingent valuation exercise can be gauged. As Hanemann (1994) points out, one method is to replicate the contingent valuation study. For this study, this was not possible. A second approach is to compare the contingent valuation results with actual behaviour. This was not possible either for this study. A third approach is to compare the contingent valuation approach with indirect methods. For this study the results of the contingent valuation approach were compared with the results of two indirect methods, namely the cost of illness and the avertive behaviour approaches, the data for which were obtained using the same questionnaire to gather the CVM bids. The cost of illness approach estimates that a farmer on average incurs a cost of around Rs 5,465 a year due to exposure to pesticides. This amounted to a little more than an average farmers monthly income which was around Rs 4,500 in 1996 (Central Bank of Sri Lanka, 1997).[16] On the other hand, the avertive behaviour approach[17] estimates show the costs to be around Rs 405 a year or about 12% of the monthly income of an average farmer per year. This is consistent with the hypothesis that contingent valuation bids exceed the sum of cost of illness and avertive behaviour expenditures (Harrington & Portney, 1987). Unfortunately, there are no other studies of WTP that have been conducted in Sri Lanka to determine the value of avoiding direct exposure to pesticides by farmers that can be compared with the results of this study. Furthermore, as regards 'content or face validity' the survey instrument was carefully designed and pre-tested, as described earlier, in order to make sure it adequately covered the domain of the goods it intended to measure. Another test of validity is the estimation of the bid curve (Hanley & Spash, 1993) that is discussed in the next section. The results show that the subsistence farmers' WTP to avoid direct exposure to pesticides increase with farmers' income, size of household, poor health resulting from direct exposure to pesticides and the length of time a farmer is involved in handling and spraying pesticides on the farm for a given year. The econometric work is discussed in more detail in the next section.

7. FACTORS INFLUENCING THE WILLINGNESS OF SUBSISTENCE FARMERS TO PAY TO AVOID DIRECT EXPOSURE TO PESTICIDES AND THE ASSOCIATED ILLNESSES

In this section the relationship between contingent valuation WTP to avoid direct exposure to pesticides affecting the health of users (farmers) and the various socioeconomic, health and time variables are examined. The aim is to determine how much of the variation in the contingent valuation WTP bids can be explained by differences in the observed characteristics. The results of the econometric analysis are relevant, not only for economic models explaining the factors affecting the demand to avoid direct exposure to pesticides but also for policy decision making. Although there are no zero bids, the Tobit analysis is used because it is the more theoretically appropriate method for WTP data sets (Halstead et al., 1991). The results are then compared with the OLS estimates.

7.1. Hypotheses about the Determinants of the Valuation Bids

For the econometric analysis, the standard socio-economic measures such as income, education, household size and age are used. The socio-economic measures selected as explanatory variables are similar to those that have been used by Brien et al. (1994) who examined the relationship between contingent valuation WTP bids and socio-economic variables for various illnesses (not pollution related). Such work has also been influenced by the theoretical work carried out by Grossman (1972) and Feldstein (1993) on demand for health and medical care. It is the perceived view that differences in demand for health and medical care can be influenced by education, age, income and other socio-economic factors. Hence, it is hypothesized that the better educated individuals are likely to bid more to avoid direct exposure to pesticides and the resulting illnesses and individuals with higher incomes are willing to pay more to avoid direct exposure to pesticides and the resulting illnesses. It is also hypothesized that although older individuals are expected to bid less than young people (because they are at the end of their working lives and hence have a need to save for retirement years), they would be willing to pay more to avoid the extra costs associated with avoidable illnesses such as those arising from exposure to pesticides as they grow older.

It is also hypothesized that individuals in bad health are expected to bid higher amounts for improvements in their health, reflecting increasing marginal disutility of bad health (Brien et al., 1994). This follows Grossman's (1972) standard assumption of diminishing marginal utility of good health, where, the more healthy days an individual experiences, the less he or she is willing to pay to obtain an additional good day. This can be shown by a marginal WTP curve for improved health. As shown in Fig. 1, the curve slopes downwards due to the fact that the individual (by assumption), is willing to pay less for a marginal increase in health if his or her health is good, than if his or her health is bad. A dummy variable is used to describe the health status of the respondents. The dummy variable indicates whether a respondent has suffered ill-health from exposure to pesticides or not. One is used to indicate 'ill-health' and zero is

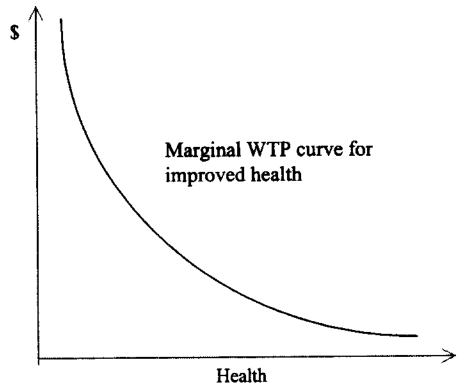

Fig. 1. Expected Relationship Between Ill-health and Marginal Willingness to Pay for Improved Health.

used to indicate 'no ill-health' from exposure to pesticides resulting from handling and spraying pesticides on the farms.

Another important variable used is the length of time pesticides are used on the farm in a given year. It is hypothesized that the more months a farmer is engaged in handling and spraying pesticides, the more likely he or she is to suffer health risks. Therefore, such an individual would bid more to avoid exposure to pesticides and hence the resulting illnesses that accompany such exposure.

7.2. Regression Analysis

Using the primary data collected from the field survey, OLS and Tobit regressions are performed. In the regression, farmers' monthly income (INC), age (AGE), education (EDU), number of household members (NOI), whether a farmer has suffered ill-health or not from exposure to pesticides (SICK) and length of time pesticides are handled and sprayed shown by the months of pesticide use (TIME) are used as explanatory variables. The dependent variable is the contingent valuation WTP bids to avoid direct exposure to pesticides. The following specification was developed for the regression analysis.

$$CV = f(\underset{+}{INC}, \underset{+}{AGE}, \underset{+}{EDU}, \underset{+}{NOI}, \underset{+}{SICK}, \underset{+}{TIME})$$

The variables in the above function are identified in Table 4 showing summary statistics. The expected signs of the partial derivatives are indicated beneath each argument in the above function.

Table 4. Means and Standard Deviations to Avoid Direct Exposure to Pesticides.

Variable	Label Description	Mean	SD
CVM	Contingent Valuation Bid	11,471.18 Rs	12684.43
INC	Yearly Income	56,978.10 Rs	53855.00
AGE	Age	40.00 Yrs	11.20
EDU	Education	7.57 Yrs	3.27
NOI	Household Size	4.72 (persons)	1.62
SICK	Sickness	0.96	2.10
TIME	Pesticide Use	8.99 (months)	2.10

7.3. Summary Statistics

Reported in Table 4 are the means and standard deviations for all variables that were included in the regression analysis. The mean contingent valuation bid is Rs 11,471 for a year which is around two and a half times the monthly income of an average farmer in Sri Lanka. The yearly average income is Rs 56,978. The mean age is 40 years and the household size is around five per family. The average amount of education is 7.5 years.

7.4. Regression Results

The results of the OLS and Tobit estimates are presented in Table 5. For the Tobit analysis, only the t-ratios are reported. Tests conducted showed evidence of violations of assumptions such as linearity, constant variance and normality of the distribution of the residuals. This was minimized by taking the logs of the dependent variables in the regression analysis. The log transformation of the dependent variable also improved the goodness of fit. The 'tolerances and variable inflation factor and the collinearity diagnostics' for the variables showed that multicollinearity was also not a problem among the independent variables. The regression results are interpreted using a one tailed test. The null hypothesis is H_0: $\beta = 0$ and the alternative hypothesis is H_1: $\beta < 0$ or H_1: $\beta > 0$.

7.5. Discussion of Results

The OLS and Tobit analyses show that there are no significant variations in the two analyses that affect the significance of the results. This may be due to the absence of non-zero values in the contingent valuation bids. The results show that income and household size of the respondents are significant factors influencing his or her WTP to avoid direct exposure to pesticides.

The education coefficient, however, is small and is insignificant. These results confirm the Brien et al. (1994) study contradicting the theoretical belief that the higher the level of education, the higher is the contingent valuation WTP bids. However, this result is not surprising because in most 'developing country' schools environmental subjects, including harmful effects of pesticides, are not taught. Hence, the level of awareness is limited. The age coefficient is also insignificant.

On the other hand, there is ample evidence to show a strong relationship between the respondents' ill-health resulting from exposure to pesticides and

Table 5. Regression Results of the Contingent Valuation Willingness to Pay bids to Avoid Direct Exposure to Pesticides.

	OLS			Tobit	
Variable	Unstandardized Coefficients B	Standardized Coefficients Beta	Standard Error	t-Ratio	z = b/s.e.
Yearly Income (INC)	3.1E-06	0.151	1.42E-06	2.190***	2.229****
Age (AGE)	–0.007	–0.077	0.007	–0.993	–1.011
Education (EDU)	0.005	0.016	0.263	0.210	0.214
Household Size (NOI)	0.097	0.142	0.047	2.061***	2.097***
Pesticide Use (TIME)	0.087	0.165	0.037	2.316****	2.357****
Sickness (SICK)	1.024	0.199	0.360	2.843****	2.894****
(Constant)	6.672	–	0.605	11.02****	11.217****

R Squared = 0.11 Adjusted R Square = 0.09 Standard Error = 1.06 F = 4.35

The asterisks *** *, ***, ** and * indicate 1, 2.5, 5 and 10% level of significance respectively for a one tailed test.

No non-zero observations

n = 203

Note: We interpret the beta coefficients in the regression results rather than the B coefficients. This is because the units of measurement of the variables are not the same. Hence, the coefficients are not directly comparable. Therefore, when variables differ substantially in units of measurement, the sheer magnitude of their coefficients does not reveal anything about their relative importance. Hence, in order to make the regression coefficients somewhat more comparable, the coefficients have been standardized to take into account the differences in the various units of measurement of the variables. Therefore, the beta coefficients are the standardized coefficients while B coefficients are the unstandardized coefficients. The standardized beta coefficients can be calculated directly from the regression coefficients using the following formula: $B_1\ (S_x/S_y)$ where B_1 is the regression coefficient and S_x is the standard deviation of the independent variable and S_y is the standard deviation of the dependent variable (SPSS, 6.0, 1993).

the bids reflecting increasing marginal disutility of illness. This variable is highly significant. The length of time a farmer is engaged in handling and spraying pesticides for a given year is also significant.

8. HEALTH POLICY IMPLICATIONS

The regression results are useful for policy decision making. The results show that incomes of farmers play a significant part in the determination of the WTP bids in avoiding direct exposure to pesticides. This is consistent with general economic theory which is applicable even to a 'low income' developing country. The size of household, too, is significant. The results also show that education and age do not play a significant part in the determination of the WTP bids, while the effects of pesticide exposure on the health of the users and the length of time pesticides are sprayed for a year, play a significant role in the determination of the WTP bids. The education variable being insignificant in the determination of WTP to avoid direct exposure to pesticides has many implications. It is known, as studies have shown, that exposure to pesticides cause many long-term illnesses, in addition to short-term health effects, most of which are incurable. The level of education here does not play a role in preventing such short-term and long-term illnesses. The problem is even more serious, especially because pesticide pollution that is released into the environment can be non-point in nature and is also very potent. The total effect of all the pesticide pollution generated by a very large number of users is even more lethal and is made more dangerous because of the pesticide stock in the environment. Furthermore, another implication that arises out of the results is that individuals begin to take note of the need to avoid direct exposure to pesticides only after they have suffered from ill-health due to direct exposure, until which time they may use pesticides. Hence the damage done is very large. By the time the victims of direct exposure to pesticides begin to pay to avoid direct exposure because of the adverse effects (ill-health), the damage done would be irreversible. Also in such a situation, the results imply that even governments would begin to act only once the damage to human health has begun to take effect and the damage done is visible. Foresight in avoiding the dangers and the health effects arising from direct exposure to pesticides does not play a role. It indicates adaptive behaviour rather than a non-myopic preventive type of behaviour.

The long-term consequences are even more frightening. We know that studies in the United States have shown a link between pesticide poisoning and long-term effects such as various cancers, loss of memory and tumors (Hoar, 1986; Nielson & Lee, 1987; Blair & Zahm, 1993). In such a case, even if a respondent realizes that a chronic illness is due to direct exposure to pesticides and is willing to pay to avoid such exposure, it would be too late since most of these illnesses are not completely curable. Such a trend is very dangerous.

This study focussed attention only on the health effects of farmers exposed to pesticides. However, it is worth mentioning some of the possible external effects that can arise from pesticide pollution on farms. Given the high use of pesticides on farms, it is possible that the effect on neighbouring individuals is likely to be considerable since water sources and the environment are affected. It is possible that the entire food chain can be affected as a result although no study has been conducted in Sri Lanka to determine this relationship. The damage done to consumers of cultivated food crops, though unknown, could also be high. It has been shown that pesticides can be taken up by crop roots and end up in the food chain. Furthermore, the residues of pesticides sprayed on crops can end up in the food harvested. The cost of other negative externalities could also be high.[18] Several interesting negative externalities arising from pesticides were noted during the field study. Herbicides used on onion plots to destroy weeds, when spread to neighbouring farms destroyed other crops which were not resistant to the herbicides used.[19] The damage incurred was very large since it affected the crop of an entire season. There were several externalities of this nature. The damage to fish production is unknown, although, in USA, Malaysia, the Philippines and Bangladesh, 'fish kills' have been linked to pesticide pollution (United States Environmental Protection Agency, 1990; Ministry of Finance, Bangladesh, 1992; Dinham, 1993).

9. CONCLUSIONS

The contingent valuation approach was used to obtain bids to avoid direct exposure to pesticides and the resulting adverse health effects among farmers in Sri Lanka. The approach considered all the costs incurred by farmers, including the intangible costs such as pain, discomfort, stress and suffering. The costs were shown to be high.

The regression results showed that a farmer's income and size of household do play a role in the determination of the WTP to avoid direct exposure to pesticides even in a 'low income' developing country, while the level of education and age do not influence a farmer's WTP to avoid direct exposure to pesticides. The results also show a strong relationship between poor health resulting from direct exposure to pesticides and the size of bids, reflecting increasing marginal disutility of illness. Furthermore, the results indicate the possibility of health decisions being reactive rather than non-myopic preventive in nature. The length of time a farmer is involved in handling and spraying pesticides for a given year is a significant variable in explaining the determination of the WTP bids to avoid direct exposure to pesticides. These have important policy implications. For example, government intervention in controlling pesticide use may be justified not only due to the adverse expenditures generated by such use but by the initial myopia which farmers display in applying pesticides. Their WTP to avoid pesticide damage increases with their experience of poor health from pesticide use. There are probably two reasons for this: (a) greater awareness of the health risks associated with pesticide use; and (b) the Grossman (1972) effect mentioned earlier. It seems likely that in the majority of cases, the likelihood of ill-health from pesticide use and the disutility from that ill-health are underestimated by farmers using pesticides. Hence, the need for some form of government intervention to protect individuals from exposure to pesticides. Finally, however, the whole issue rests in the principle of caveat emptor, that the buyer/user alone is responsible. For this purpose educating the farmers about the hazards of using pesticides at current high levels without adequate precautions is of utmost importance.

NOTES

1. No dis-aggregated data are available from the National Poisons Centre. Bed head tickets of all the government hospitals have to be examined for this task to isolate cases resulting from direct exposure to pesticides from handling and spraying on the farms. However, the bed head tickets of a selected number of hospitals in the study area were examined and it was found that a considerable number of cases were due to occupational poisoning, although the majority of cases were due to self-ingestion (suicides).
2. An illness is described as serious where the respondent was hospitalized, a moderate illness is where the respondent took treatment from a physician but was not hospitalized and a mild case is where a respondent was neither hospitalized nor sought treatment, but took home-made self-treatment and incurred other private costs.
3. A distinction between private and public costs is made because government hospital treatment is free of charge in Sri Lanka. However, certain prescriptions may have to be purchased from a pharmacy and laboratory tests may have to be conducted in a private clinic. Furthermore, some farmers also seek treatment from private clinics.
4. Hanemann (1994) argues that the CVM was originally proposed by Ciriacy-Wantrup (1947).
5. Carson (per com., 1998) states that more than 2,500 studies have been carried out in more than 50 countries using this method. A literature survey conducted for this work found that more than 100 studies have been carried out in developing countries alone using this technique during the last decade.
6. For a complete review of studies carried out in the health care field up to the mid 1990s, see Donaldson (1993); Johansson (1995) and Diener et al. (1998).

7. Loomis and Duvair (1993) point out that "the payment of higher taxes is not an emotionally neutral subject for many people and that such a payment vehicle may increase the number of protest bids" (p. 288).

8. For a contingent valuation study that uses the wording 'higher prices' to describe the payment, see Ready et al. (1996). Also see Kenkel et al. (1994).

9. The WTP bids can also be obtained by a series of questions confronting them with different prices for the program depending on their previous answers or it can take the form of a dichotomous choice or close-ended question format where respondents are told how much each would have to pay if the measure passed and are then asked to cast a simple "yes" or "no" vote. This dichotomous choice contingent valuation question format has gained popularity over the last few years. It is also a NOAA (1993) panel recommendation. This is due primarily to their purported advantages in avoiding many of the biases known to be inherent in other formats used in the contingent valuation method. However, there are many disadvantages too. For example, Whittington (1998) points out that if the amount the enumerator asks lacks credibility, the respondent is unlikely to answer the question on the basis of the prices asked. Furthermore, the type of format depends on the nature of the study and conditions prevailing in developing countries. These considerations were taken into account in selecting an open-ended question format for this study. For a detailed discussion on the disadvantages of the dichotomous choice format and the advantages of the open-ended question format for this study, see Wilson (1998).

10. It must be mentioned here that it would have been very useful to interview farmers who do not currently use pesticides. This would have been useful to compare the differences in the willingness to pay bids between those who use pesticides and those who do not. For example, Liu et al. (1996) compare the WTP amounts residents of urban areas are willing to pay to avoid a recent episode of an illness with WTP bids given by residents who live near large petrochemical complexes. They show that the WTP to avoid an illness recurrence of the most recent episode is three times larger for those living in areas near the petrochemical complexes than for those living in urban areas. However, for this study this was not possible because of the difficulty in locating farmers who were not using pesticides.

11. The exchange rate prevailing during the study period (June–September, 1996) was \$1 = Rs 55 (approximately).

12. Small-scale farmers cannot afford to buy inputs in bulk. Hence, the reason for purchasing small quantities of pesticides and other inputs from time to time. They also have limited storage facilities.

13. See Cowan and Gunby (1996) for reasons why IPM has been slow to be adopted on farms.

14. For a discussion on the environmental and human costs of pesticide use, see Wilson (2000); Wilson and Tisdell (forthcoming).

15. Jeyaratnam et al. (1982), too, use these survey data for their study.

16. It is interesting to note that the daily wage rate paid for pesticide sprayers in Sri Lanka is higher than the daily wage paid for other farming activities (Central Bank of Sri Lanka, 1997). This reflects the risks involved in spraying pesticides.

17. This method is also known as the defensive behaviour approach.

18. In the field study undertaken to gather data, the costs of negative externalities were not considered.

19. It is interesting to note that the Pea plant in recent years has been genetically engineered for the purpose of making it completely immune to herbicides such as Roundup. This enables farmers to blitz the entire farm with Roundup which virtually kills all plants and weeds (and also other micro-organisms and insects) except the cultivated Pea crop.

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12. A COMPARISON OF POLICIES TO REDUCE PESTICIDE POISONING COMBINING ECONOMIC AND TOXICOLOGICAL DATA

David Sunding and Joshua Zivin

ABSTRACT

The paper combines health risk generation and economic models to compare the efficiency of the two main types of policies to reduce worker injury from toxic substances: occupational safety regulations designed to limit exposure *to toxic substances and taxes and bans that are intended to reduce* contamination, *or total use of the product. The model is developed with reference to pesticide poisoning of farm workers. General conditions for the relative efficiency of policies are derived. Empirical results indicate that protective clothing requirements achieve given reductions in poisonings with about half the cost of a pesticide tax. Implications of the model for regulating other types of worker injury from hazardous inputs are presented.*

INTRODUCTION

Worker injury resulting from the use of toxic substances is a serious public health problem, and is of significant concern to workers and labor unions. In the period 1994 to 1998, workers reported over 27,500 injuries annually

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resulting from exposure to toxic inputs in private industry (Bureau of Labor Statistics, 1999). These injuries can be quite serious, with symptoms ranging from skin irritation and nausea in mild cases to paralysis and death in severe cases.

Governments employ a variety of policies to reduce workplace injuries resulting from exposure to toxic substances. One common remedy is occupational safety regulations that establish guidelines for the proper use of toxic materials in the workplace. These requirements include product labeling, mandated use of protective clothing, and adoption of specialized use equipment. Another strategy for reducing worker injury from toxic substances is to reduce the use of the hazardous product itself. Lowering use may be accomplished by financial incentives such as risk-based taxation that change relative prices, or by more extreme measures such as a ban on the use of a particular chemical.

This paper combines a toxicological model of the health risk generation process with an economic model of input demand to compare the efficiency of policies to reduce worker injury from toxic substances. The health risk model is based on an explicit consideration of three fundamental risk factors: contamination, exposure and dose-response. This framework is standard in public health, but has been underutilized in policy analysis. Our paper considers the impact of various policies on each of these risk factors, and combines effects to measure the total reduction in worker injury. The economic model of toxic input demand is used to assess the responsiveness of firms to changes in policy, and to measure economic impacts. The integrated health risk-economic model is used to compare the efficiency of the two main types of policies to reduce workplace injury from toxic substances: occupational safety regulations that are designed to limit worker *exposure* to toxic substances to input taxes and bans that are intended to reduce *contamination*, or total use of the product.

The model is developed with reference to pesticide poisoning of farm laborers. Pesticide poisoning of farm workers is a serious problem nationwide, and has been addressed by federal and state policies, including the Farm Worker Protection Act. In California alone, there were 448 reported incidents of work-related pesticide poisonings among agricultural workers in 1998 (California Environmental Protection Agency, 1999). The reality is almost certainly worse than these figures indicate, as farm workers are widely known to under-report injuries (Coye, 1985). Given the number and seriousness of farmworker poisonings, and the wide variety of public policies used to improve worker health, it is of interest to consider the relative efficiency of these interventions.

In the next section, we introduce the health risk model and discuss the essential risk factors. We develop a conceptual model of pesticide use and worker exposure to pesticides, and identify the marginal impacts of various worker protection regulations. The model is then applied to a case study of insecticide use in California's lettuce industry. We measure both the worker health benefits and economic costs of representative policies. In particular, we consider the impacts of a pesticide tax. This policy has been advocated by a number of economists and pesticide policy analysts as an efficient way to regulate pesticide injury to farm workers, consumers and the environment. The other policies considered are mandated use of protective clothing and specialized equipment that are typical of occupational safety regulations in agriculture as well as other industries.

HEALTH RISK AND PUBLIC POLICY

The magnitude of health risk in a population of workers is defined as the probability of manifesting a physical ailment as a result of contact with a toxic substance used in the workplace. If there is a pre-existing incidence of this ailment in the population, then risk refers to the incremental risk incurred by human exposure to the contaminant. Health risk is represented as the product of three fundamental risk factors: (i) ambient contamination; (ii) human exposure; and (iii) the manifestation of physical symptoms (termed the dose-response relationship):

$$R(I) = C(I) \cdot E(I) \cdot D, \tag{1}$$

where I indexes government policy. This multiplicative formulation is reasonable when the marginal risk is relatively small, and is particularly germane to cases where there is pre-existing background risk (Van Ryzin, 1980; Crump & Howe, 1984; Krewski & Van Ryzin, 1981). Note that since the dose-response function is simply a biological mapping from exposure to illness, it is not affected by policy choices.

One common type of policy used to improve worker health is occupational safety regulations that require employers to post warning signs, train workers how to safely handle toxic materials, and invest in specialized equipment and protective clothing. Pesticide policy provides a good example of these types of regulations. Federal regulations require farmers to post signs on the perimeter of their fields warning workers that chemicals have been recently applied. Farmers must also offer classes demonstrating how workers can avoid unnecessary contact with pesticides, and what to do in case of such contact; these training classes typically involve hiring a registered instructor and conducting a short series of on-site meetings. Federal and state regulations also mandate protective clothing that varies by job type. For example, pesticide applicators need highly specialized gear, while harvest workers are simply advised to wear hats, long-legged pants, and long-sleeve shirts. All of these requirements affect exposure levels in that they minimize human contact with pesticides after they have been applied. To the extent that farmers incur marginal costs as a result of these regulations, they may also change the incentives for pesticide application and thereby reduce contamination as well.

Another type of health-improvement policy frequently advocated by pesticide policy analysts is taxation of hazardous substances. This policy provides a marginal disincentive for the producer to use the toxic material, thereby reducing workplace injury. In terms of the risk factors outlined in (1), such an input tax reduces contamination but leaves exposure unaffected.

THE MODEL

We now develop a more detailed model of contamination and exposure with reference to pesticide poisoning of farm laborers, and show how these health risk factors are affected by changes in government policy.

Contamination

Given our interest in farm worker health, we define contamination as the amount of pesticide available for human contact. Contamination per acre is defined as the product of the share of acres treated with the pesticide (RS), the amount of chemical used per acre treated (a), and a crop-specific coefficient (v) that converts per-acre application into a deposited residue. Formally,

$$C(I) = RS(I) \cdot a \cdot v. \tag{2}$$

The application share is related to the elasticity of demand for the pesticide. The total number of pounds of the pesticide used per acre is $a \cdot RS(I)$. Thus, the elasticity of demand is equal to the elasticity of RS with respect to the per-acre cost of pesticide application, denoted as m. The pesticide demand curve will be employed later to measure the change in profit resulting from a change in m.

The coefficients a and v are exogenous. The amount of pesticide applied per acre, a, is specified on the product label, typically in terms of pounds of active ingredient per acre. The coefficient v maps pounds of chemical per acre into micrograms per square centimeter of surface area available for worker contact. As such, this coefficient will vary by job type. For mixer/loader/applicators, the application amount corresponds exactly with the amount available for worker contact (i.e. $v = 1$). For harvest workers, this coefficient will be less than one, reflecting the fact that only a fraction of all applied pesticides end up as deposited foliar residue available for contact (Zweig et al., 1985). The exact value of v is heavily influenced by application method and the shape and size of the plant.

Exposure

Farm worker exposure to pesticides is determined by job type, the length of time between application and human contact with the residue, and by government regulations. Let H be the date that the worker makes contact with the pesticide (i.e. the harvest date in the case of a harvest worker) and let A represent the date of pesticide application. Thus, $H - A$ is the period of time over which the pesticide will decay. Exposure is given as

$$E(I) = q(I) \cdot e^{-k[H-A]}, \tag{3}$$

where k is a residue decay parameter, and q is an exposure coefficient. The decay parameter is a function of the type of pesticide applied, and may also be influenced by weather conditions such as humidity, temperature and rainfall.

The coefficient q is in part a function of the type of job performed by the worker. Job type influences the intensity and duration of contact with hazardous substances. For example, pesticide applicators have more exposure to pesticides than harvest workers or installers of irrigation equipment. The coefficient is also heavily influenced by workplace safety measures. We will detail some of these influences in the empirical section.

Health Impacts of Contamination and Exposure Regulations

It is now possible to examine the impact of various worker protection policies on health risks. Combining Eqs (1), (2) and (3) it follows that the health impact of the policy change is

$$\frac{\partial R}{\partial I} = (\eta \cdot \Delta m + \Delta q) \cdot R, \tag{4}$$

where η is the pesticide demand elasticity,

$$\delta m = \frac{dm/dI}{m} \text{ and } \Delta q = \frac{dq/dI}{q}.$$

Recall that this expression assumes a given exposure coefficient q, and is thus conditional on job and crop type. The first term in the parentheses captures the effect of the policy change on pesticide contamination, and the second term is its effect on exposure.

Examination of expression (7) reveals a main advantage of our approach: only basic information is needed to assess the impact of policy reforms on worker health risks. Specifically, one must know the responsiveness of contamination and exposure to changes in regulations. Pesticide demand elasticity measures are readily available, from the manufacturer's marketing studies, from academic studies or by construction as in the method described below.[1] Relatively little public health information is needed to assess the change in health risk resulting from pesticide policies. In particular, it is necessary to ascertain the impact of policies on the exposure coefficient q. This coefficient is familiar to public health specialists familiar with pesticide use, and can often be obtained with little difficulty.

A pesticide tax unambiguously reduces the expected number of pesticide poisonings for all job types since it reduces the incentives for pesticide application and does not affect exposure. Other government policies also affect the marginal benefits from pesticide taxation. Public policies that reduce the variability of per-acre returns (e.g. yield or revenue insurance) reduce the marginal incentives to apply pesticides, and thereby increase the marginal reduction in poisonings from a pesticide tax. Similarly, market institutions such as the ability to forward contract will lower the marginal impact of a pesticide tax.

While economists in particular seem to favor pesticide taxes as a means to reduce poisonings, regulators more often implement outright bans. Pesticide bans can be easily modeled in this framework; conceptually, they are akin to large pesticide taxes in that they reduce contamination, lower the level of expected profits and leave exposure unaffected. Occupational safety regulations regarding clothing, equipment, posting and training affect worker health risk by reducing exposure. They may also reduce contamination by increasing the marginal cost of pesticide application. The effects of these regulations on exposure and contamination are reinforcing, and the net effect of strengthening occupational safety regulations is an unambiguous reduction in health risk.

It is instructive to compare policies in terms of lost profit per poisoning averted. The change in per-acre profit per unit change in R is

$$\left.\frac{d\Pi}{dR}\right|_i = \frac{(1 - 0.5 \cdot \eta \cdot \Delta m_i) \cdot dm_i \cdot RS}{\sum_{j \in J} (\eta \cdot \Delta m_i + \Delta q_i^j)}$$

where $i \in I$ indexes the policy change and $j \in J$ indexes job type.[2] A pesticide tax leaves the exposure parameter unaffected, while an occupational safety requirement alters both m and q. Which of these policies reduces worker poisoning with the smallest reduction in profit per poisoning averted, is an open, empirical question depending on the relative changes in m and q. In the following section, we demonstrate how expression (5) can be applied to a typical situation to measure the relative effectiveness of alternative worker protection policies.

It is also important to compare the economic welfare cost, or "deadweight loss," of the policies. The deadweight loss from the regulation is

$$\left.\frac{dW}{dR}\right|_i = \frac{0.5 \cdot dm_i \cdot \eta \cdot \Delta m_i \cdot RS}{\sum_{i \in J} (\eta \cdot \Delta m_i + \Delta q_i^j)}. \tag{6}$$

This expression assumes that tax revenues are fully and costlessly recycled (i.e. are simply a transfer from farmers to some other unspecified group), and have no welfare implications beyond the deadweight loss associated with raising them. Equivalently, the expression is predicated on the assumption that protective equipment is produced at a constant marginal cost of zero. These assumptions are extreme, and most likely the true welfare impact of the policies will lie between that calculated in (6), and the profit loss measured by (5) that does not net out tax revenues or equipment expenditures.

CASE STUDY

The case considered in this section is mevinphos applied to leaf lettuce in California's Salinas Valley. Mevinphos (2-carbomethoxy-1-methylvinyl dimethyl phosphate) is an organophosphate insecticide used on a variety of crops, mainly vegetables. In California, close to half of all mevinphos used is applied to lettuce; other major uses are on cauliflower, broccoli and celery. It is used primarily as a short-residual foliar insecticide to "clean up" crops just prior to harvest. Mevinphos is used primarily to control aphids, although it is also effective against mites, grasshoppers, cutworms, leafhopper caterpillars and other insects. The pesticidal activity of mevinphos is due to its inhibition of acetylcholinesterase activity.[3]

There are numerous reported cases of worker poisoning involving mevinphos, most resulting from acute exposure. Indeed, mevinphos is responsible for more acute illness than any other insecticide currently in use (California Environmental Protection Agency, 1999). The principal symptoms of mevinphos poisoning are nausea, diarrhea, vomiting, pinpoint pupils, tremors and, in extreme cases, paralysis. Mevinphos is not known to be carcinogenic, and is not believed to cause reproductive or developmental toxicity. In California, there were 548 reported cases of acute mevinphos poisoning involving farm workers during the period 1982 to 1991. There were 68 cases involving one or more days of hospitalization and 201 cases involving one or more lost sick days during this period (O'Malley, 1992).

Contamination

At present, the share of leaf lettuce acres treated with mevinphos in the Salinas Valley is 0.751 (California Environmental Protection Agency, 1995). This value is taken as the pre-reform level of RS. The cost of mevinphos application is $50 per acre, which includes application expenditures and the cost of the chemical itself (University of California Cooperative Extension, 1992). The application rate, a, is set at 0.25 pounds of active ingredient per acre, as specified on the product label (California Environmental Protection Agency, 1994). The crop-specific contamination coefficient v is taken from Formoli. It is customary to report contamination per acre treated as $a \cdot v$; this coefficient is listed in Table 1, along with the application share (RS).

The demand elasticity for mevinphos used on leaf lettuce in the region is needed to assess the change in profit following a change in pesticide regulation that affects the cost of application. In the case study, we set the long-run elasticity at –1.760 (Sunding). In other applications where there is no demand elasticity readily available, it can be obtained by estimation, or by construction from the output demand elasticity.

Table 1. Baseline Parameters for Harvesters and Mixer/Loader/Applicators.

Type of Worker	RS	$a \cdot v$	q	$\exp\{-k \cdot PHI\}$	D	Worker Days per Acre	Total Cases
Harvest	0.751	0.066	20.136	0.604	3.85E-05	60	26.465
MLA	0.751	2.802	1.321	1	3.85E-05	0.114	0.232

Exposure

We now turn to a description of the data used to assess worker exposure to mevinphos. We consider two types of workers: pesticide mixer/loader/applicators (MLAs) and harvest workers. These two job categories have the highest incidence of pesticide poisonings, and are thus of natural interest.[4] As discussed earlier, exposure is based on values of q and k, and the length of time between application and exposure. In theory, the decay parameter k can be affected by weather conditions such as rainfall and temperature (Spear et al., 1977; Nigg et al., 1978). However, Spencer et al. have demonstrated that for many vegetable crops produced under varying weather conditions in California, the mevinphos decay coefficient is in fact constant at $k=-0.72$ μg/cm^2/day.

Another important factor determining exposure is the length of time between contamination and human contact, $H-A$. In the case of a pesticide applicator, this value is zero since exposure occurs at the time of application. For harvest workers, this value corresponds to the mandated reentry or pre-harvest interval, whichever is longer. The pre-harvest interval for mevinphos used on lettuce in Monterey County is 7 days (California Environmental Protection Agency, 1994).

The exposure parameter q is also affected by government regulations. Formally,

$$q=\frac{k_D\cdot t}{bw} \tag{7}$$

The parameter k_D is a crop-specific dosing coefficient, measured in cm^2/hr, which relates the amount of active ingredient contacted to an hourly dose of poison. This coefficient varies by task and also according to protective clothing and equipment regulations. Typically, this dosing coefficient is broken down into a transfer component and an absorption component. Transfer components are often expressed as a clothing penetration measure or as an inhalation uptake, depending on the route of contact, and vary by type of clothing and equipment (see for example Fong et al., 1990; Maddy et al., 1981; Brodberg & Sanborn, 1996). Absorption coefficients are generally extrapolated from toxicological studies conducted on laboratory animals, are specific to neither clothing nor equipment, and vary by route of exposure. Thus, workplace safety measures such as clothing and equipment regulations affect q through its impact on the transfer component of the dosing coefficient. The variable t denotes the time of exposure and the variable bw denotes bodyweight. As is standard in the toxicological literature on risk assessment, these variables are assumed to take the values 8 hours and 70 kilograms, respectively.

Baseline Worker Health Risk

In order to assess the health impacts of pesticide use on harvest workers, we must discuss the relevant contamination and exposure measurements for this type of worker. As mentioned earlier, the approved application rate for mevinphos is 0.25 lbs. of active ingredient per acre. Because we are analyzing exposure for harvest workers, however, we are interested in the fraction of this applied active ingredient that is deposited on the plant, $a \cdot v$. This fraction is known as deposited foliar residue and at the approved application rate is equal to 0.066 μg of active ingredient per cm^2 of plant surface area (Spencer et al., 1991).

Pesticide decay and protective clothing and equipment conditions harvest worker exposure to this level of contamination. As mentioned earlier, the pre-harvest interval for mevinphos used on lettuce in Monterey County is 7 days (California Environmental Protection Agency, 1994). Currently, there are no protective clothing requirements for leaf lettuce harvest workers. We assume that workers wear the customary outfit of a long-sleeved shirt, long-legged pants, shoes, and a hat. This outfit results in a dosing coefficient, k_D, equal to 176.19 cm^2/hr, which in turn yields an exposure parameter, q, that equals 20.136 cm^2/kg (Brodberg & Sanborn, 1996; Spencer et al., 1991). Combining the figures on foliar residue (contamination) with those for exposure we can derive an average daily dose (ADD), where

$$ADD = RS \cdot a \cdot v \cdot q \cdot e^{-k(H-A)}. \tag{8}$$

In our case study, ADD is equal to 0.603 μg/kg/day. All that remains is to multiply this daily dose by the dose-response parameter and the total person-hours spent at harvesting leaf lettuce. The dose-response relationship, D, is a biological relationship that maps contaminant dose into a manifestation of clinical symptoms. Typically, this relationship is determined by extrapolating from animal models, or by reference to actual illness. Based on clinical observations of mevinphos poisoning from O'Malley, we set D at 3.85×10^{-6}. A 60-person crew can harvest one acre of leaf lettuce in one 8-hour workday (University of California Cooperative Extension, 1992). Finally, there are 19,000 acres of leaf lettuce under cultivation in the Salinas Valley (Monterey County Water Resources Agency, 1995). Thus, it follows that our baseline number of mevinphos poisonings of leaf lettuce harvest workers is 26.465 per year in the Salinas Valley.

We now turn to MLAs. These workers come into contact with the raw pesticide product. Thus, the relevant contamination measurement for this group of workers is simply the full application rate per acre (i.e. $v = 1$), so that $av = 2.802\ \mu g/cm^2$. No time elapses between application and exposure, suggesting an important role for workplace safety measures. Ground mixer/loader/applicators of pesticides (MLAs) are subject to strict state and federal protective clothing and equipment requirements. These requirements include: long-sleeved shirt, long-legged pants, coveralls or a rainsuit, chemical-resistant gloves (during mixing and loading only), boots, a hat, a respirator, and a face-shield or goggles. Although these requirements may seem onerous, MLA poisonings in California and nationwide still occur.

Estimates by Formoli et al. suggest a dosing coefficient equal to 11.554 cm^2/hr ($q = 1.321$), which yields an average daily dose of 2.779 μg/kg/day. One ground mixer/loader/applicator can apply mevinphos to 8.8 acres in one 8-hour workday (University of California Cooperative Extension, 1992), and, as before, we assume that in a typical year there are 19,000 cultivated acres of leaf lettuce in the Salinas Valley. Using the same dose-response parameter as for harvest workers, it follows that total baseline MLA poisonings given these protective clothing requirements is 0.232 cases per year for leaf lettuce produced in the study region.

Policy Impacts

In our case study, we examine three distinct policy interventions. As a means of protecting harvest workers we analyze the requirement of vinyl gloves. This policy was chosen due to the high frequency with which harvest workers' hands contact pesticide residue, and the proclivity of mevinphos to be absorbed dermally.[5] For MLAs, recent research suggests that some reduction in the risk of pesticide poisoning may be obtained through the use of closed-cab equipment while applying pesticides (Alcoser & Formoli et al., 1992). Accordingly, we also model the impacts of mandating the use of closed-cab rigs while applying mevinphos. To the extent that these policies reduce pesticide use, they will effect workers not directly targeted by the intervention. For example, if vinyl gloves reduce pesticide use, then MLAs benefit through the reduction in contamination levels. Similarly, harvest workers may enjoy reductions in the risk of pesticide poisoning if regulators require closed-cab rigs, even though their own working conditions do not change.

Lastly, we consider the effects of a pesticide tax, as many economists have suggested that risk-based taxation is an attractive way to regulate poisonings resulting from pesticide use. As shown previously, pesticide taxes reduce the incidence of worker poisonings by lowering the amount of pesticides applied by farmers. We analyze the marginal impact on worker health risk of a tax on mevinphos.

As discussed earlier, polices that require workplace safety clothing or equipment reduce worker poisoning by reducing both contamination and exposure. The cost of vinyl gloves are based on the assumption that each worker will wear two pairs of gloves per day, yielding 120 gloves per acre. The price of vinyl gloves is \$8.75 per 100, so a glove requirement adds \$9.50 per acre to the cost of mevinphos application (Gempler's, 1999). Given our pesticide demand elasticity, this additional cost provides a marginal disincentive to use the pesticide. The additional cost is significant in percentage terms, and reduces the share of acres treated with mevinphos by 33.44% (from 75.12% of leaf lettuce acres in the Salinas Valley to 50.00%). These results are presented in Table 2.

The glove policy reduces exposure by limiting the contact between a harvest worker's hands and the chemical. Results from toxicological field studies indicate that the donning of vinyl gloves while harvesting reducing the exposure dosing coefficient, kD, to 119.28 cm^2/hr, so the exposure parameter, q, is 13.63 cm^2/kg (Spencer et al., 1991). Combining the contamination and

Table 2. Policy Impacts.

Policy	*RS*	*q*	Poisoning Cases	Lost Profit and Welfare per Poisoning Averted (\$/poisoning)
Baseline				
Harvest	0.751	20.136	26.465	NA
MLA	0.751	1.321	0.232	NA
Gloves				
Harvest	0.500	13.630	11.924	7,732; 1,551
MLA	0.500	1.321	0.154	
Closed-Cab				
Harvest	0.717	20.1360	25.260	14,697; 344
MLA	0.717	1.249	0.209	
Tax				
Harvest	0.725	20.136	25.535	14,921; 267
MLA	0.725	1.321	0.224	

exposure effects, it follows that the vinyl glove requirement reduces the number of harvest worker poisonings from a baseline of 26.465 to 11.924 annually.

Through its effect on the cost of mevinphos use, the glove requirement also reduces the number of MLA poisonings by lowering contamination. In the baseline, there are 0.232 MLA poisonings annually; the glove requirement reduces this to 0.154 cases each year. This change is proportional to the reduction in mevinphos use since MLA exposure conditions are unchanged.

Utilizing the demand relationship established for mevinphos together with these measures of the change in poisonings, it is possible to calculate lost profit per poisoning averted. Following Eq. (5), lost profit amounts to $7,772 per poisoning prevented by the vinyl glove requirement. From (6), lost welfare is $1,551 per case.

A closed-cab requirement for MLAs reduces poisonings in a manner analogous to the glove requirement. The annualized cost of the closed-cab regulation is $1.30/acre (John Deere), which lowers the share of acres treated with mevinphos to 0.717. In addition, the closed-cab requirement reduces MLA exposure by reducing the dosing coefficient on worker exposure, k_D, to 10.930 cm^2/hr, so that q is now equal to 1.249 cm^2/kg (Formoli et al., 1993). Thus, the closed-cab regulation results in 0.023 fewer MLA poisonings annually. Due to the reduction in mevinphos contamination caused by the cost increase, the number if harvest worker poisonings decreases by 1.205 per year as a result of the closed-cab requirement. Interestingly, the absolute effect of the closed-cab requirement is greater for harvest workers than for the MLAs whom the regulation is designed to protect. Again, the regulation reduces profit, as shown in Table 2. In the case of the closed-cab regulation, this loss amounts to $14,772 annually for every poisoning averted. Lost welfare is smaller for this policy. From Eq. (6), it follows that the deadweight loss of the closed-cab requirement is $344 per case. As will become clear shortly, the result that the closed-cab requirement is more efficient than the glove requirement follows from the size of the change in the per-acre cost of the pesticide.

Finally, a pesticide tax reduces worker health risk solely through its influence on the amount of pesticide use; worker exposure per unit of pesticide applied is unaffected. Table 3 shows the reduction in welfare per poisoning averted for various levels of the tax. One result of this analysis is that the welfare cost of the tax increases with the size of the tax. This result holds because of two factors. First, the ratio of deadweight loss to tax revenues increases as the tax increases; second, the marginal impact of a tax increase on the number of poisonings is invariant with respect to the level of the tax under the linear demand specification implicit in (6). Thus, the welfare cost of reducing the number of poisonings via taxation of the pesticide varies considerably from

Table 3. Welfare Impacts of a Tax.

Tax ($/acre)	Lost W per Poisoning Averted ($/case)
1	267
2	535
3	802
4	1,069
5	1,337
6	1,604
7	1,871
8	2,138
9	2,406
10	2,673

$267 per case for a unit tax per acre to nearly $1,337 per poisoning for a $5 tax.

It is instructive to compare the welfare cost of a tax to the welfare cost of the occupational safety regulations. Consider first the vinyl glove requirement. Setting the tax at $9.50 per acre (or $38 per pound of active ingredient), it follows that the welfare cost of the tax is $2,539 per poisoning prevented, while the glove requirement reduces welfare by $1,551 per case. Thus, by this measure, the glove policy is more efficient than the pesticide tax. Similar calculations reveal that the closed cab regulation is also more efficient than the pesticide tax.

With equivalent assumptions about recycling of tax revenues and equipment expenditures, the model points out that the welfare cost of a pesticide tax is greater than the welfare cost of occupational safety regulations, measured in terms of lost welfare per poisoning averted. This result follows from the fact that occupational safety regulations reduce both contamination and exposure, whereas a pesticide tax only reduces pesticide use (e.g. contamination). That is, for any arbitrary pesticide tax, one can (in principle) find an occupational safety regulation with the same cost per acre that also reduces exposure. Because the cost increase is the same, the policies reduce contamination by the same amount. Yet, the occupational safety regulation also reduces exposure, thereby achieving a greater reduction in worker health risk than the tax.

It is worth noting that this result does not run counter to economists' usual advocacy of price instruments to control environmental health risks. The problem here is not with price instruments per se, but rather that a pesticide tax targets one factor determining health risk – pesticide use – when the ultimate public policy goal is to reduce health risk itself. A policy like workers' compensation (or even tort liability) that raises the cost of a poisoning, rather than an input to poisoning, is better designed to deal with the problem. We discuss these factors in more detail below.

Discussion

In the case study, policies directed at exposure are more efficient than policies aiming to reduce the amount of pesticide use. Indeed, the vinyl glove requirement is twice as efficient as the pesticide tax. Of course, the relative efficiency of contamination- and exposure-related policies is an empirical question. In different settings, policies will have different effects on profit and worker exposure.

Other biological factors can alter the analysis. For example, there may be long-term risks (e.g. cumulative effects) in addition to the short-term risks captured in this paper, and there may be risks to workers from the interaction of different pesticides. Addressing the question of long-term risks involves a different health risk generation model than the acute poisoning framework adopted here (for details, see Van Ryzin, 1980). Interaction effects are more problematic. The U.S. Environmental Protection Agency is struggling with this problem as it implements the Food Quality Protection Act of 1996. Little is known about how pesticide risks aggregate, even across pesticides within a single chemical class such as organophosphates. The problem here is not that the farmer will switch from one pesticide to another in response to a change in relative prices; this effect can be dealt with by considering a net change in health risk and profit using the model in this paper. Rather, the difficulty arises in that the farmer may use other pesticides for other purposes, and that workers may be exposed to a variety of pesticides as they work on other crops in other locations. Unfortunately, it is not possible to treat these issues comprehensively at this time as much basic toxicological research remains to be done.

It is of interest to compare the lost profits resulting from the regulations to the value of the benefits received from reductions in the number of worker poisonings. Between 5 and 10% of all organophosphate poisonings are characterized as severe. A severe case of organophosphate poisoning requires 3 days of hospitalization, with the first day spent in intensive care, plus 80 hours of lost work. The cost of one day in an ICU is $3,000 (including medications and monitoring), and the cost for a day in a normal hospital room is $800 (SPARC). At typical agricultural rates, 80 hours of lost work costs roughly $1,000. These costs alone amount to $5,600, counting neither the worker's discomfort, nor the cost of future health problems, and may grossly underestimate the true benefit of preventing a poisoning. A moderate case of organophosphate poisoning requires an office visit to a physician and medications, which amount to roughly $500 if the visit is to a physician's office, or $200 if the visit is to a clinic. Also, the worker typically misses 2 days of work, so that lost wages are roughly $200. Thus, the minimum cost of a moderate case of meivnphos poisoning is $400 to $700 per case.

The results of the case study raise some observations about optimal combinations of policies to reduce worker health risk. In particular, while the case study found that exposure controls were relatively efficient, it does not follow that they should be the only policies employed. The glove requirement, for example, reduces worker health risk at a welfare cost of roughly $1,500 per poisoning averted. If the social value of reducing organophosphate poisonings exceeds $1,500, then other interventions are desirable. While exposure controls such as the vinyl glove requirement may be relatively efficient, it is also true that they are limited. In the case study, the glove requirement reduces mevinphos poisonings from 26.697 to 12.078 cases, but it cannot reduce poisonings any further. One advantage of the pesticide tax is that it can effectively eliminate risk from the pesticide taxed by reducing its use to zero.

Given the limitations inherent in various occupational safety regulations, it is worth considering how regulations may be combined to achieve arbitrary reductions in the incidence of worker poisonings. It is cost-effective to "stack" regulations in terms of their cost to producers, starting with the most efficient. Consider the following example. Suppose that a regulator wishes to achieve a 75% reduction in the number of mevinphos poisonings. Starting at the baseline, the closed cab requirement reduces poisonings to 25.463 cases, which is still less than 75% (6.674). Next, suppose that the regulator imposes the vinyl glove requirement on top of the closed cab requirement. Now, the baseline cost of mevinphos application is $51.30 per acre, and adding $9.50 per acre due to the glove regulation reduces the share of acres treated with mevinphos to 0.483. The number of poisonings is reduced to 11.664, still above the level desired. Imposing a pesticide tax of $14.78 per acre on top of these occupational safety regulations achieves the remainder of the reduction.

The total welfare cost of this three-part regulation is $62,474, which amounts to $3,120 per poisoning averted. The per-case welfare loss is well above that for any of the exposure or tax interventions alone. This result follows from the fact that the marginal impact of a tax increase on the number of poisonings decreases as the exposure parameter decreases. Thus, the welfare loss from taxation per poisoning averted is higher after imposition of the glove and closed-cab regulations ($8,209 per case averted) than before. For comparison, note that if the 75% reduction in the number of mevinphos poisonings were achieved with a pesticide tax alone, the tax would need to be set at $21.30 per acre. The total cost of this policy would be $5,696 per poisoning averted, which is significantly above the cost of the sequential, or combination strategy.

An important question raised by the case study results is the effectiveness of various liability rules for worker injury from pesticide use. The pesticide tax only gives farmers an incentive to reduce contamination by reducing pesticide use. As shown above, this may not be the most efficient way to reduce worker health risk. A policy that makes farmers monetarily liable for worker injury is more efficient in that it would give them an incentive to adopt exposure control technologies whose cost is less than the value of the avoided injury to workers.

It should be noted that current workers' compensation system does not give farmers an adequate incentive to adopt efficient exposure controls, despite the fact that workers' compensation insurance premiums are a significant part of total labor expenses in agriculture (Rosenberg, 1992). Premiums are determined by three factors: the type of work performed by the employees covered, total payroll, and the claims record of the policyholder over the trailing three-year period. The program does give farmers some incentive to prevent pesticide poisoning since premiums increase with the number and severity of injuries. However, the program falls short of the optimal due to several factors. Workers are widely known to under-report injury (Coye, 1985), and not seek medical attention even when it is called for.[6] More important, many pesticide injuries can be latent or caused by interactions of chemicals applied by the numerous farmers for whom the worker may be employed. In this sense, pesticide injury has much in common with non-point source pollution; injury is measurable (even if under-reported), but it is hard, if not impossible, to establish a proximate cause for the harm or identify a specific grower responsible for the injury. Thus, policies similar to ambient taxes, or other interventions, may work to efficiently reduce pesticide poisonings. This topic should be the subject of more intensive research.

CONCLUSIONS

This paper considers the relative effectiveness of two types of policies to reduce pesticide poisoning of farm laborers: occupational safety regulations that reduce exposure and policies like pesticide taxes or bans that reduce ambient contamination. The paper develops and implements a model to measure the level of pesticide poisoning of farm workers and the economic cost to farmers and others from various policies. The conceptual model considers the incidence of poisoning as the product of three factors: ambient contamination, human exposure and the dose-response relationship. Our analytic results show that it is important to consider the impact of public policies on each of these components separately, and to combine effects to measure the full impact of worker protection policies on health risks and producer welfare. For example, one of the most important health benefits of exposure controls is that they also reduce contamination by raising the cost of pesticide application.

A key feature of our method is that it can be implemented with basic economic and toxicological information. Models such as ours can help to open lines of communication between public policy analysts and public health specialists working in the area of occupational safety. The data needed from toxicologists and epidemiologists to measure marginal worker health and farmer welfare impacts are expressed in the model in terms that are meaningful to them, such as dosing coefficients and rates of decay.

The paper presents a detailed empirical example that compares the health impacts and costs of three policies to reduce the incidence of pesticide poisoning of lettuce workers in California. Occupational safety regulations requiring harvest workers to wear vinyl gloves and mandating the use of closed-cab application equipment are relatively efficient means of reducing worker health risks. In particular, the vinyl glove requirement is roughly twice as efficient as the pesticide tax, measured from the perspective of lost profit per poisoning averted. This finding casts doubt on the efficiency of policies, like taxation or hazardous inputs, that control workplace injury by altering relative prices.

The basic health risk framework developed here can be applied to a wide variety of problems, with appropriate modifications. Through demand studies, much is known about the marginal productivity of toxic substances in a variety of industry settings. Similarly, toxicologists and industrial hygienists have worked for years to characterize risks from toxic substances in many settings. These are the basic components of the health risk model; our research indicates how they can be combined to improve the efficiency of public policies designed to reduce worker health risk.

NOTES

1. We assume that farmers do not alter the application rate, a, because doing so relieves the chemical manufacturer of all legal liability.
2. The per-acre loss in profit is

$$\Delta PS = \int_{m}^{m+dm} D(RS)dRS.$$

Taking a linear approximation to this integral, it follows that $\Delta PS = (1 - 0.5 \cdot \eta \cdot \Delta m) \cdot dm \cdot RS$. This expression assumes that wages and workers' compensation premiums are unaffected by changes in health risk. This approach is consistent with the observations that pesticide injuries are under-reported, latent and the result of exposure on numerous farms.

3. Cholinesterases are a family of enzymes found throughout the body that hydrolize choline esters. In the nervous system, acetylcholinesterase (AChE) is involved in the termination of impulses across nerve synapses including neuromuscular junctions by rapidly hydrolizing the neural transmitter, acetylcholine. Inhibition of AChE results in overstimulation followed by depression or paralysis of the cholinergic nerves throughout the central and peripheral nervous systems (California Environmental Protection Agency, 1999).
4. Sometimes, MLA work is performed by the farm owner.
5. This policy is explored in California Environmental Protection Agency, 1994.
6. Pesticide injuries can sometimes mimic the symptoms of more common ailments (e.g. headache, nausea and diarrhea).

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