Physical and Economic Model Integration for Measurement of the Environmental Impacts of Agricultural Chemical Use

John M. Antle and Susan M. Capalbo

Chemical use in agriculture has, over the last fifty years, been the good, the bad, and the uncertain. By all accounts, agricultural productivity increased significantly in the last half century due in part to the introduction and expanded use of agricultural chemicals. More recently, however, some agricultural practices, including increased chemical use, are viewed as having a major impact on the larger ecosystem and as being an important source of environmental nonpoint pollution. Recent groundwater-monitoring programs in the United States sponsored by the U.S. Environmental Protection Agency (EPA) have revealed contamination of underground water supplies by pesticides, nitrates, and other industrial organic chemicals. It has also been questioned whether the high rates of productivity growth that have characterized modern agriculture can be sustained with technologies that disrupt the ecosystem. While there is no doubt that production agriculture does affect the ecosystem, it is not obvious that these effects are necessarily widespread and deleterious to the environment or to long-run productivity, or, if they are, how they should be valued and traded-off with other social objectives.

The overall concern regarding environment quality and the possibility of long-run improvements in both environmental quality and agricultural productivity has lead to heightened interest in research that incorporates environmental and health impacts into evaluation of the social benefits and social costs of agricultural technologies and policies. There are at least two reasons to explain why analysts have tended not to include the environmental and health impacts in their analyses of returns to agricultural research or in their evaluation of specific policies or programs: deficiencies in methodology and data. On the methodology issue, a comprehensive analytical framework is needed that combines field-level relationships among management practices, environmental attributes of the farmland, and nonpoint pollution with impacts on human health. The research from various disciplines (physical, biological, economic, and health sciences) needs to be integrated into an analytical framework that, to be useful for policy analysis, makes the link between the physical changes in environmental and resource quality attributable to agricultural practices and the valuation attached to the changes in environmental quality and the subsequent impacts on human health. With respect to data deficiencies, the concerns are in two related areas: (1) the information needed to quantify the environmental quality and agricultural production relationships has generally not been available and (2) the data on human-health effects of exposure to agricultural chemicals are far from complete.

This paper begins to address these deficiencies, first by developing an approach to integrating disciplinary research to quantify and value the impacts of agricultural chemical use, and second by highlighting the data requirements for this research. The basic premise of this paper is that economics provides a framework to integrate the disciplinary models and data for policy analysis. Economists must play a central role in the organization and interpretation of the physical, biological, health, and social science research that is needed to quantify and value the impacts of agricultural chemicals. The approach is illustrated using the issue of chemical contamination of surface water and groundwater.

The paper is organized as follows. The first section includes a discussion of the conceptual framework for evaluating the net social benefits of chemical

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use. In this section, we argue that modern benefit-cost analysis provides the conceptual framework in which to organize a coherent approach to the problem of incorporating environmental and health costs into public-policy analysis and for addressing the uncertainties inherent in this type of analysis. The second section presents an overview of the characteristics of the physical models that can be used to predict the movement of chemicals in soils and discusses a prototype model for assessing pesticide concentrations in the soil and groundwater. In the third section, attention is focused on modification of economic production models. The last section addresses the methodological issues that arise in integrating physical and economic models for use in the benefit-cost framework.

Benefit-Cost Analysis of Chemical Impacts on the Environment

Benefit-cost analysis (BCA) provides the framework for assessing the impacts of agricultural chemicals. An ex ante BCA of a change in regulatory policy or a change in technology requires prediction of the direct and indirect effects, expression of the effects in terms of common units, and a determination of the net impact on social welfare. The BCA process can be presented as a sequence of steps as shown in Figure 1, which uses the example of pesticides. The first step, denoted by box 1, is to determine the effect of the policy or the change in technology on the output and input decisions of the farmers who utilize the pesticide in their production practices.

The second step is to quantify how a farmer's response affects the magnitude of the benefits and costs. In the case of a pesticide-use reduction, changes in environmental contamination, food residues, and occupational exposure give rise to the benefits (see boxes 2A, 2B, and 2C); the effects on production and resource use determine the costs (box 3).

The environmental impacts of changes in chemical use depend on the physical processes of chemical transport through soil and water mediums (box 4) and subsequent contamination of secondary food sources. These physical processes include degradation of the pesticide by soil microorganisms, chemical degradation, pesticide sorption by mineral and organic constituents in the soil, plant uptake of pesticide materials, volatilization, and the effects of water-flow processes that disperse and distribute the pesticides during passage through the unsaturated soil zones. Analysis of the effects of changes in pesticide use on human health involves both human and environmental risk assessment (see boxes 5 and 6).

The third step in the BCA process is to express the benefits and costs in a common unit that reflects their valuation by the affected individuals. These values are commonly expressed in monetary units. The valuation of the costs of the pesticide-use restrictions or changes in production technology requires information on production changes, market prices, and the impacts on consumers caused by changes in food production and prices. These costs can be measured as changes in producer and consumer surpluses or related welfare measures (box 7). The valuation of the benefits is much more controversial; it involves predicting the impacts on the environment and estimating nonmarket values.

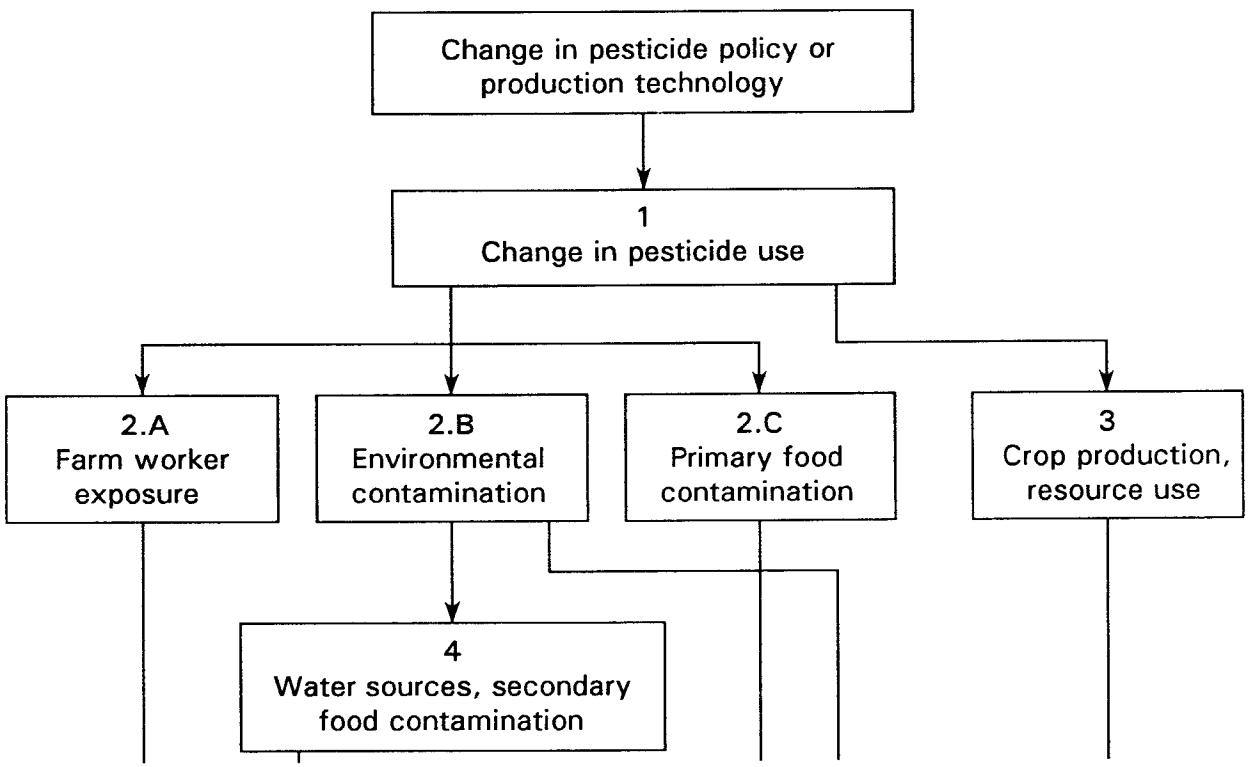
The final step in the BCA process is the determination of the net impact on social welfare. This requires a criterion for determining what qualifies as an increase in welfare and a means for aggregating the impacts that may occur at different points in time. The standard procedure is to use the present discounted value of net benefits as the criterion for evaluating changes in social welfare.

Physical Models for Quantifying Contamination Levels

Physical models for quantifying chemical-pollution externalities need to address movement of chemicals to both surface water and groundwater. In the last three decades, an extensive literature has been generated by research aiming to trace the movement of surface-water contaminants. Climate, watershed and soil characteristics, and crop management practices have been found to affect the magnitudes of the impacts (see Jury, Focht, and Farmer).

Concern over groundwater contamination is a relatively recent development, and, as a result, models that predict chemical leaching to groundwater are less developed than models that predict chemical runoff to surface water. To model potential loadings to groundwater, a model is needed to trace the movement of the chemical from the application site down through the unsaturated (or aerated) zone and into the saturated zone. The saturated zone is the area in which all the void spaces are filled with water; in the unsaturated zone, the void spaces are filled with both air and water, the proportion of which is important in modeling transport rates. The soil, or root zone, is typically a part of the unsaturated zone.

The fate of a pesticide or other chemical applied to soil depends largely on two of the pesticide's


Change in pesticide policy or production technology
1
Change in pesticide use
2.A
Farm worker exposure
2.B
Environmental contamination
2.C
Primary food contamination
3
Crop production, resource use
4
Water sources, secondary food contamination

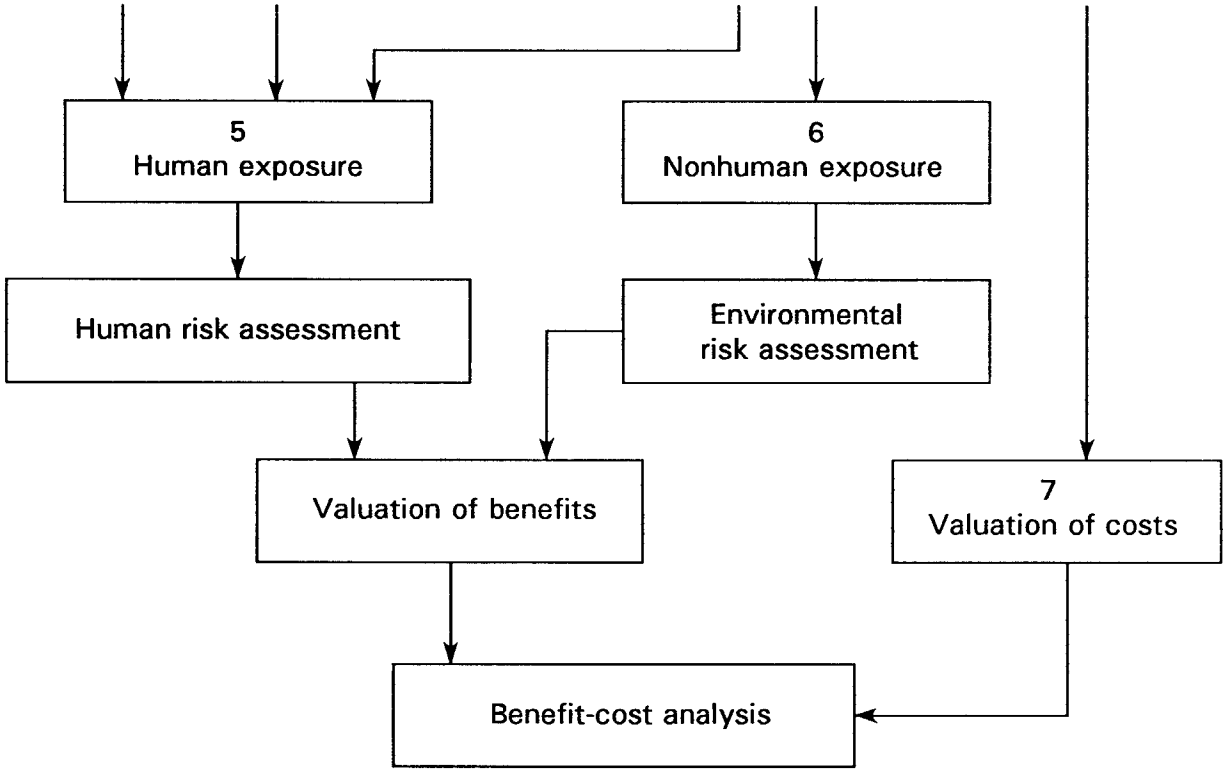


Figure 1. Major Components of a Benefit-Cost Analysis of a Change in Pesticide Use

properties: persistence and solubility. Persistence is a measure of a chemical's rate of degradation and is usually measured in terms of a chemical's half-life. Solubility, sorption, and volatility determine how a compound partitions among water, soil, and air phases and affect whether the chemical is moved primarily with sediment or water. When a pesticide is applied, some of it will adhere to the organic carbon in the soil particles; this is called adsorption. Some of the pesticide will mix with soil water and move down with the soil water. An inverse relationship exists between the solubility of the pesticide and its sorption to soil. A partition coefficient value is used to describe the ratio of pesticide concentration in the adsorbed phase and the solution phase. The smaller the partition coefficient, the greater the concentration of pesticide in solution. Hydrologists have noted that the greatest threat to groundwater through leaching is associated with a pesticide with a small partition coefficient and a long half-life.

Chemical Transport Models: An Overview

Although the specific structures of the chemical fate and transport models vary, most models contain some standard components. These include the following components.

1. *Surface runoff generation component* describes the transformation of precipitation into runoff. The soil surface and profile provide major controls on the response of the surface-water system. During interstorm periods, pesticides may be applied and undergo a variety of transformation and degradation processes affecting the total mass of each constituent available for entrainment and transport. Land-use practices such as tillage affect the infiltration, runoff, and erosion processes. The processes composing the surface-runoff system are hydrology, sediment, nutrients, and pesticides. (A detailed presentation of modeling surface runoff is provided by Beasley, Pionke, and Leonard.) The U.S. Department of Agriculture Soil Conservation Service Curve Number (SCSCN) model is commonly used to estimate runoff. This method relates direct runoff to daily rainfall as a function of a curve number representing soil type, soil drainage properties, crop type, and management practice.

2. *Soil and groundwater component* describes chemical movement through the unsaturated soil zone and may also describe movement into the saturated zone. Not all models trace the movement of chemicals through the unsaturated zone to the saturated zone.

3. *Erosion component* estimates soil loss due to erosion. This is important when determining potential for groundwater contamination because soil sediment is a medium of transport for adsorbed pesticides. A pesticide or nutrient that is transported off the field via eroded soil is not available for leaching to groundwater. The Universal Soil Loss Equation (USLE), or a modification of the USLE, is frequently used to model erosion. The USLE accounts for factors such as rainfall, crop management, slope conditions, and erosion-control practices in calculating soil loss per acre.

4. *Soil adsorption and desorption component* estimates the partitioning of a chemical between adsorbed particles and dissolved chemicals. This component estimates what portion of the chemical may be transported by soil sediment and what portion may be transported by soil water. It may also model volatilization and decay of the chemical.

Chemical-transport models can be divided into three broad categories: research models, screening models, and management models (Wagenet and Rao provide a detailed discussion of these models). Research models provide quantitative estimates of water and solute movement but usually involve extensive data demands on the system to be simulated. Management models are less data-intensive and less quantitative in their ability to predict water and solute movement under various environmental conditions. Although most managerial and research models are field-scale models, Wagenet and Rao indicate that there has been limited field testing of either the research or management models to date, and thus little attention has been focused on the so-called management models for the actual purpose of managing pesticide or fertilizer usage. The existing research models are useful for management purposes only if computer facilities and time are virtually unlimited.

Screening models are used to evaluate and compare pesticide fate and transport under alternative environmental conditions. The screening models have relatively low data demands and are designed to be relatively easy and inexpensive to use. One useful output of these models is to categorize chemicals into broad behavioral classes. These models have relevance in the pesticide registration process, where the properties of a pesticide that has not been field-tested can be inferred from the class in which it is placed. Several simple indexes useful to screen and rank pesticides in terms of their potential to leach into groundwater have been developed by Rao, Hornsby, and Jessup. These ranking schemes are based on a screening model that determines the relative travel time needed for the pesticide to migrate through the unsaturated zone and the relative mass emissions (loadings) from the unsaturated zone into the groundwater.

Jury et al. have also developed a screening model of the pesticide-leaching process. This model relaxes the uniform first-order decay assumption for pesticide degradation in the unsaturated zone, which characterizes the Rao et al. model, and replaces it with a biochemical decay relationship that decreases with soil depth. The results of both screening models indicate a significant dependence on site-specific soil and environmental conditions, suggesting that these factors, as well as the pesticide properties, need to be taken into account when screening for groundwater-pollution potential.

Wagenet and Rao caution against using existing screening models to predict environmental changes. They indicate that the recent interest in using models to predict the fate of pesticides in water and soils has provided an impetus to improve upon the accuracy of both screening and research models. One of the most promising avenues to proceed for developing policy models is condensing the comprehensive description provided by research models. Examples of such an approach are the recent changes to the PRZM and LEACHM models (see Wagenet and Hutson) and the Jury et al. model and the prototype model discussed in the following section.

A Simple Pesticide-Leaching Model

One major disadvantage of the large-scale research simulation models is their lack of attention to the movement of chemicals through the unsaturated zone, although groundwater components have recently been appended to some models. A second disadvantage of these models is simply the size and data requirements. Most utilize daily and often hourly climate data to simulate chemical movement.

As an alternative, researchers have been developing screening models to evaluate pesticide groundwater-pollution potential (Jury et al. and Rao et al.). This approach is promising for use in regulatory BCA, and thus we illustrate the integration of such a model into the net-benefit specification.

Two key variables in assessing the behavior of chemicals as they leach into groundwater are pesticide residence time and the fraction of the pesticide remaining as functions of depth in the unsaturated zone. Physical relationships can be used to estimate residence time, t_i, and the time required for a pesticide particle to travel from land surface to the depth of interest, z_i, as a function of physical parameters such as water flux per unit surface area; residual moisture content; dry-bulk density; the organic-carbon partition coefficient of the pesticide; and the percentage of organic carbon in the layer.

The fraction of the pesticide remaining at the depth of interest is calculated taking into account both the decay and root-uptake processes. The fraction of the pesticide that remains after decay that occurs during its transport through each soil layer can be calculated by solving the equation for irreversible first-order reactions allowing for the known half-life of the pesticide:

$$r_i = e^{-0.693\ (t_i/h_i)}, \tag{1}$$

where r_i denotes the fraction of the pesticide remaining after transport in the ith layer; t_i denotes the time of travel (residence time) in the layer of interest, in days; and h_i denotes the half-life of the pesticide in the layer, in days.

These latter values are assigned to each layer in the system based on empirically obtained figures from field and laboratory experiments. The percentage of the original pesticide applied to the land surface that remains after transport through more than one layer is the product of the values of r_i for each layer. The percentage of the pesticide remaining after transport and decay through all layers is then

$$r_z = \prod_i r_i. \tag{2}$$

The key parameters in determining the amount that remains are generally half-life of the chemical, porosity, partition coefficient (which is determined by the organic-carbon coefficient of the pesticide and the percentage of organic carbon in each layer), water flux, and water content.

The root-uptake process also must be estimated and, as a first-order approximation, can be assumed to be proportional to the root uptake of water evapotranspiration. To obtain the fraction of the pesticide remaining after these two processes (root uptake and decay) have occurred, the amount of pesticide remaining after decay is multiplied by the ratio of the amount of water flux at the depth of interest to the amount of water entering the ground at land surface:

$$C(X_j) = (r_z)(q/w)(X_j), \tag{3}$$

where q denotes water flux per unit surface area, X_j denotes the amount of pesticide applied, and w denotes the rate at which water enters the ground. Equation (3) could be incorporated into a net-benefit analysis, as illustrated in the fourth section of this paper.

To utilize this kind of model, information would be required on soil (physical) and pesticide characteristics. The soil characteristics include the rate at which water enters the ground, the rate of deep percolation below roots, the thickness of the root zone, the depth to the water table, and the density of solid matter in the unsaturated zone. Other layer-specific physical characteristics include the type of

material, the residual moisture (water) content, the porosity, and the organic-carbon content of the soil. Pesticide characteristics of importance to these models are organic-carbon partition coefficient and the half-life in each layer. In addition, data on pesticide applications are also needed. Of the above information, only the pesticide-application levels and the amount of water entering the ground at time of application would need to be collected on a regular basis. Presumably the other soil and pesticide characteristics are not time variant, but do vary spatially.

Environmental Exposure Modeling

More general approaches to environmental-quality modeling are also being developed. The standard approach to modeling environmental exposure is to assume that chemicals are distributed into various environmental compartments as functions of chemical properties, environmental factors, and chemical use according to equilibrium partitioning models (Mackay et al.). For example, it may be assumed that a pesticide applied to a field will be partitioned among air, water, soil, flora, and fauna. Symbolically,

$$C_{ij} = C_i(X_j, K_{ij}, E_i),$$

where C_{ij} is the concentration of the jth chemical in the ith partition; X_j is chemical use; K_{ij} is the partition coefficient; and E_i is a vector of environmental factors.

The environmental contamination in each partition can be translated into exposure of the kth species through the expression

$$e_{jk} = \Sigma_i C_{ij} A_{ijk} (\gamma),$$

where e_{jk} is the exposure of the kth species to the jth chemical; $A_{ijk}(\gamma)$ is the rate of uptake of the jth chemical in the ith partition by the kth species; and γ is a vector of individual species characteristics.

Thus, in general, total exposure of the kth species to the jth chemical is a function $e_{jk}(X, K, E, \gamma)$, where the arguments are vectors of chemicals used, partition coefficients, environmental characteristics, and species characteristics. These exposure measurements can in turn be valued and used as a basis for BCA.

Economic Production Models

The economic behavior of agricultural firms can be represented as a two-level decision process corresponding to the short run and the long run (Figure 2). In the short run, firms make production decisions regarding outputs (types of crops and allocation of acreage among crops) and variable inputs (such as labor hours, fertilizer applications), taking as given the available technology and the existing stocks of physical capital and other resources used in production. These short-run decisions may be important in the analysis of externalities because they may include the use of agricultural chemicals that are a source of pollution. In the long run, firms make investment decisions based on their expectations of future market conditions, technology, and resource availability. Their long-run decisions include the total acreage of the farm operation and the quantities of physical capital employed. The long-run decisions may also have important consequences for externality generation. For example, the choice of tillage method (conventional tillage versus reduced or no-till) may have an impact on soil erosion and herbicide use, and hence on pollution caused by chemical runoff.

Producer Behavior in Static Models

Farmers typically are assumed to be concerned with the private benefits and costs of their farm operations and thus do not take into account the longer-term impacts of their production activities on the ecosystem or on human health caused by agricultural pollution that occurs off their farms. Farmers' longer-term capital-investment decisions are not likely to have a direct impact on the generation of externalities. Under these circumstances, for analysis of externalities it is appropriate to model the short-run behavior of farmers as the maximization of the economic returns to management and capital, taking technology, management, and capital investment as given in each period. The analysis using a static model thus focuses on the output and input decisions that are made in each production period, given technological, economic, and resource constraints. For the measurement of externalities, the effects of the output and input decisions on physical-resource stocks and living organisms in the ecosystem can in turn be measured. To measure the sequence of externalities generated over time, the biological system's changes can be incorporated into the economic model to define the resource constraints on production in the next period, and the analysis can be repeated.

The short-run economic behavior of an agricultural producer can be modeled in terms of profit maximization; more generally, risk management and other objectives can be introduced, but as a first-order approximation, profit maximization is a useful starting point. Conventional economic analysis of the profit-maximizing firm is based on the

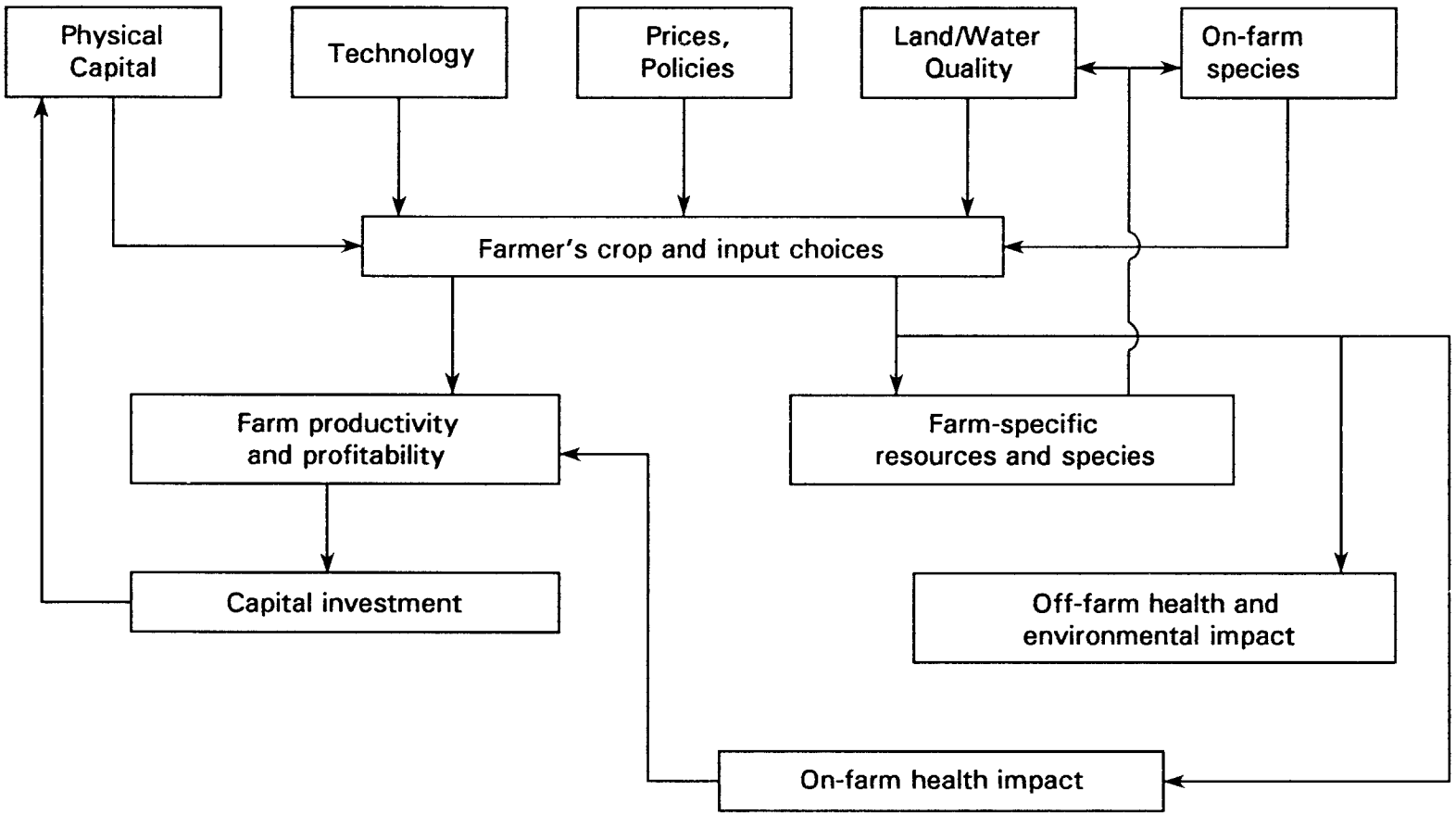


Figure 2. Production Model Linkages to On-Farm Resources and Off-Farm Environmental and Health Impacts

representation of the production process using the production function

$$Q_t = f(X_t, Z_t, \tau_t, R_t, S_t),$$

where Q_t is the maximum rate of output that can be produced in period t with variable inputs X_t (generally a vector measuring labor, fertilizer, pesticides, etc.), fixed (capital) inputs Z_t (a vector measuring land, structures, machinery and tools, etc.), and parameter τ_t, representing the state of the technology (traditional seed variety versus modern seed variety, for example). The role of physical and biological resources in the production process is represented by the vectors R_t (physical resources) and S_t (living organisms) in the production function. The vector R_t could measure physical attributes of the resources used in production, such as soil and water quality, and the vector S_t could measure populations of pests and natural enemies to pests.

The profit-maximization problem is represented as

$$\max \pi_t = P_t f(X_t, Z_t, \tau_t, R_t, S_t) - W_t X_t,$$

where P_t is the price of output and W_t is a vector of prices corresponding to the elements of X_t.

By assuming that the production function is concave in the variable inputs X_t, modern economic theory utilizes a duality correspondence between the above *primal* maximization problem and the *dual* function known as the restricted profit function:

$$\pi_t = \pi[P_t, W_t, Z_t, R_t, S_t, \tau_t].$$

This function is defined as the maximum profit the firm can earn, given P_t, W_t, Z_t, τ_t, R_t, and S_t, by choosing levels of output and variable inputs X_t. A property of the profit function is that the firm's profit-maximizing output, say Q^*, and its profit-maximizing input vector, say X^*, satisfy the following relationship:

$$\begin{aligned} Q_t^* &= \partial\pi[P_t, W_t, Z_t, R_t, S_t, \tau_t]/\partial P_t \\ &= Q^*\,[P_t, W_t, Z_t, R_t, S_t, \tau_t], \\ X_t^* &= -\partial\pi[P_t, W_t, Z_t, R_t, S_t, \tau_t]/\partial W_t \\ &= X^*\,[P_t, W_t, Z_t, R_t, S_t, \tau_t]. \end{aligned}$$

The complete production model is represented by the system of the three previous equations. Since the first equation measures short-run profit, it can be interpreted as measuring the producer surplus (net returns) used in benefit-cost analysis. For example, if a new seed variety were introduced but prices, physical capital, and resource stocks were constant, the profit function would indicate the resulting change in producer surplus attributable to the new seed variety. The equation system also shows that the introduction of the new seed variety would generally have an effect on supply of output and on the demand for inputs. The introduction of a new variety would affect the demand for agricultural chemicals. This change in the use of agricultural chemicals would provide the link from the economic behavior of the farmers to the physical and biological models used to quantify pollution externalities.

The production model also shows that, generally, the economic relationships in period t depend on the resource stocks and living organisms represented by R_t and S_t. The economic model does not determine these variables in the current production period; rather, R_t and S_t play the role of constraints on the production process. The values of R_{t+1} and S_{t+1} in the next period are determined in part by the production decisions in period t. Thus the physical, biological, and economic sectors of the model interact dynamically according to the particular structure and parameterization of the systems of equations used to represent them. Given estimates of the parameters of these equations, initial values of the stocks R_t and S_t, and predictions of the "forcing variables" such as prices that are determined outside of the model, the system of equations can be used to generate predictions of the time paths of agricultural production (Q_t), input use (X_t), and the physical and biological stocks (R_t and S_t).

Long-Run Dynamic Investment Models

In some cases it is not appropriate to use a short-run static production model to analyze externality generation. A long-run model may be needed for a variety of reasons: because the choice of capital stock is important in the amount of externality created; because farmers do take externalities into account in their decision making; or for long-run regional analysis of externality creation where the effect of the externality feeds back into the production process. To illustrate, consider a model in which physical capital evolves over time according to

$$Z_{t+1} = (1 - \delta)\, Z_t + V_t,$$

where δ is the rate of capital depreciation and V_t is the rate of gross investment each period. Similarly, assume that the dynamics of the resources R_t and species S_t are given by

$$\begin{aligned} R_{t+1} &= H(R_t, X_t, Z_t) \text{ and} \\ S_{t+1} &= B(S_t, X_t, Z_t, R_t). \end{aligned}$$

The long-run maximization problem of the farmer is now defined as choosing the sequence of investments to maximize the present discounted value of profit from each period over the relevant planning horizon:

$$\underset{\{V_t\}}{\text{Max}} \sum_{t=1}^{T} \eta_t \{\pi[P_t, W_t, Z_t, R_t, S_t, \tau_t] - U_t V_t\} + J[Z_{T+1}, R_{T+1}, S_{T+1}],$$

subject to

$$\begin{aligned} Z_{t+1} &= (1 - \delta)Z_t + V_t, \\ S_{t+1} &= B(S_t, X_t, Z_t, R_t), \\ R_{t+1} &= H(R_t, X_t, Z_t), \end{aligned}$$

where η_t is a discount factor depending on the rate of interest, U_t is the price of investment goods, and J measures the terminal value of the physical capital and resource stocks.

The above problem can be solved using optimal-control or dynamic-programming techniques. For example, the solution can be obtained by maximizing the following Hamiltonian equation:

$$\begin{aligned} H_t = \eta_t \{\pi[P_t, Q_t, Z_t, R_t, S_t, \tau_t] - U_t V_t\} \\ + \lambda_t \{(1 - \delta)Z_t + V_t\} + \mu_t B(S_t, X_t, Z_t, R_t) \\ + \rho_t H(R_t, X_t, Z_t), \end{aligned}$$

where λ_t, μ_t, and ρ_t are the multipliers for Z_t, S_t, and R_t, and represent the marginal capital values of these stocks. Maximizing the Hamiltonian and solving the resulting set of first-order conditions along with the constraints of the maximization problem gives an investment demand equation of the form

$$V_t = V[Z_t, S_t, R_t, P^t, W^t, \tau^t, U^t, \lambda_{T+1}, \mu_{T+1}, \rho_{T+1}],$$

where $P^t = (P_t, P_{t+1}, \ldots, P_T)$ and similar notation applies to other variables. Thus, the optimal investment in each period is a function of the current stocks of capital and resources, current and future prices, and the terminal values of the capital and resource stocks.

Using the investment demand equation for V_t together with the equations of motion for R_t and S_t and the equation for output supply and input demand, one can solve for the long-run paths of all variables determined by the farmer. Note that the short-run and long-run models suggest a very different model of interaction between the economic, physical, and biological models. With the short-run economic model, economic decisions are made given the states of the physical and biological variables, and the physical and biological models are solved given the behavior of farmers. Time paths for the variables in each model are obtained by sequentially solving each model and using its results to condition the solution of the other models. In contrast, in the dynamic economic model, economic decisions are made taking into account the dynamics of the physical resource stocks and the population dynamics of species. Thus, the time paths for the economic, physical, and biological variables are determined jointly in the solution of the dynamic economic model.

Model Integration

Methodological Issues

Several methodological issues arise as the physical and economic model components are brought together into an integrated model. Successful integration requires compatible mathematical structures for numerical models, and consistent statistical criteria need to be developed. In addition, several conceptual differences in model approaches exist across disciplines that need to be taken into consideration. The most important point to be emphasized in conducting this integration is the need for communication across disciplinary lines.

Physical versus behavioral modeling. First, there is a conceptual difference between the physical modeling, which relies upon physical constants, and behavioral models based on the assumed optimizing behavior of people. The structure of a physical model is invariant to changes in government policy, for example, but a model of farmer behavior may need to take into consideration the way farmers form expectations about policy. Consequently, the structure of a behavioral model may change over time as policy and other parameters change. The change in the structure of the behavioral model may in turn alter the linkages between the physical and economic models.

Experimental versus nonexperimental data. The physical and biological sciences rely primarily on data generated by controlled experiments. Economic analysis is generally based on nonexperimental data. Econometrics is devoted to the modification of classical statistical analysis so that valid inferences can be drawn from nonexperimental data. The differences in statistical methods need to be reconciled in the design of data surveys and research methodologies.

Modeling approaches. Various disciplines find particular mathematical structures to be appropriate for their problems. For models to be integrated across disciplines, all disciplinary model components must be consistent with the ultimate goal of linking the models for policy analysis.

Selecting the unit of analysis: The aggregation problem. A basic methodological problem arises in any attempt to integrate the physical, health, and economic model components into a coherent whole; each component relates to a particular unit of analysis, each of which is generally different from the unit of analysis on which cost-benefit analysis should be based. The solution to this problem is to provide a statistical representation of the integrated model that can be defined over a common unit of analysis, and then to statistically aggregate to the unit of measurement meaningful to cost-benefit analysis.

A Statistical Approach to Model Integration

A key factor that needs to be taken into account in the modeling methodology is the heterogeneity of the physical environment and the related heterogeneity of agricultural production practices and associated environmental and health effects of those practices (Antle and Just). For example, an analysis of environmental fate of a pesticide based on a set of partition coefficients may be reasonable for a well-defined physical unit—say, 100 square meters of surface area—over which a specific set of parameters and input data are valid. However, such a unit is generally much smaller than the economic or geophysical unit of analysis relevant to the assessment of social costs of chemical use. The relevant unit of analysis for social-cost assessment may be as small as a farm or as large as an entire regional watershed.

To address the heterogeneity problem, an aggregate unit of analysis can be defined as a function of the problem context; for example, for water quality problems, the unit of analysis may be the land contained in a particular watershed. The land in the aggregate unit of analysis can, in turn, be disaggregated into sufficiently small units over which a valid set of physical and economic data and parameters can be defined. Let us call such a unit a plot. Associated with each plot is a vector of physical characteristics represented by ω, which may include physical characteristics such as depth to groundwater on the plot, the partition coefficients for the plot, the slope and elevation of the plot, and so forth. A stylized physical model can then be written $C(X, \omega)$, where C is a vector of contaminant levels associated with the environmental partitions in the model (e.g., soil, air, water) and X is a vector of chemical applications.

As shown in the third section in this paper, a farmer's chemical-use decisions are functions $X(P, \psi, \tau, \omega)$, where P represents prices of outputs and inputs, ψ represents policy parameters, τ represents technology parameters, and ω is as defined above. Let the environmental characteristics of each plot of land in the region be fixed at a point of time and distributed across plots according to a distribution defined by a parameter θ. This distribution of environmental attributes induces a joint distribution for input use X, crop production Q, and contamination levels. Define this joint distribution as $\phi(Q, X, C \mid P, \psi, \tau, \theta)$.

Statistical Aggregation

The joint distribution ϕ provides a basis for statistical aggregation across the plots into quantities that can be used to conduct policy analysis at the aggregate level. For example, by integrating X and Q out of ϕ, a marginal distribution of contamination can be defined: $\phi(C \mid P, \psi, \tau, \theta)$. Using this distribution, the trade-offs between, say, mean chemical use and groundwater contamination can be estimated. This information can be combined with valuation data to estimate the value associated with groundwater contamination. In addition, an aggregate pollution function can be obtained by taking the expectation of C with respect to this marginal distribution and that relationship can be used for analysis of pollution policy (see Antle and Just for further discussion of these procedures).

To illustrate the statistical aggregation procedures, let X and ω follow a log-normal distribution such that

$$\begin{bmatrix} \ln X \\ \ln \omega \end{bmatrix} \sim N[\mu, \Sigma \mid P, \psi, \theta],$$

where μ is a (2×1) vector of means and Σ is a (2×2) covariance matrix. It follows that C is a random variable and its mean and variance are functions of μ and Σ, which are in turn functions of P, ψ, and θ. Thus, for example, the population mean contamination level may be expressed as a function of the population mean level of chemical use. This relationship can be employed in policy analysis. For example, if a dollar value could be attached to a specified reduction in environmental contamination, these data could be used in cost-benefit analyses of policies to reduce pesticide use.

A Simple Economic-Physical Groundwater Contamination Model for Policy Analysis

This section describes an integrated economic-physical groundwater contamination model for policy analysis. The model is defined for a given chemical at a given location, such as a plot or field, that is homogenous with respect to both physical and economic characteristics. It is based upon the

models presented in sections two and three of this paper.

A physical model. Following our earlier notation, let X be the quantity of chemical; C be the concentration of chemical X in groundwater; z equal the depth to groundwater; m equal the time for transport from surface to groundwater; r equal the fraction of chemical remaining after transport to groundwater; t equal time period, $t = 0, 1, 2, \ldots$; h equal the half-life of chemical in groundwater; and h^* equal $0.693/h$. Following the simple pesticide-leaching model presented earlier, assume the chemical does not move laterally in the soil or groundwater; it degrades according to first-order irreversible reactions; and the groundwater is uncontaminated at time $t = 0$. Then

$$C_t = \sum_{k=1}^{t} X_k R_{kt}, \tag{4}$$

where

$$R_{kt} = r \exp\{h^* [t - (m + k)]\}, \text{ if } t - (m + k) > 0$$
$$R_{kt} = 0, \text{ if } t - (m + k) < 0.$$

Note that R_{kt} is interpreted as the fraction remaining at time $t > k$ from application at k, including the effects of transport to groundwater and decay in the groundwater. Equation (4) is quite general and compatible with any specification of the coefficients R_{kt}. For example, R_{kt} could be specified more generally to embody the effects of lateral movement of groundwater.

An "economic" interpretation of equation (4) is possible. Since $R_{k,(t+s)} = R_{kt} \exp(h^*s)$, and $R_{k,t+s} = 0$ for $s < m$, C_t can also be expressed as

$$C_t = \exp\{h^*(m + 1)\} C_{t-1} + x_{t-m} R_{t-m,t}.$$

Thus, C_t can be expressed in the form of an equation of motion of a capital stock, $K_t = (1 - \delta) K_{t-1} + I_t$, where K_t is the stock, δ is the depreciation rate of the stock, and I_t is gross investment. Under this interpretation, $\exp\{h^*(m + 1)\}$ represents the depreciation of the "stock" of contamination due to the decay of the chemical that is already in the groundwater, and $X_{t-m} R_{t-m,t}$ represents the gross investment, which in this model is the additional chemical that was applied at time $t - m$ and leaches to the groundwater at time t.

An economic model. To illustrate the basic economic relationships, assume the simplest possible conditions: production of a single crop Q with a single variable input, the chemical X, on the given unit of land. The farmer chooses X to maximize profit π subject to the production process

$$Q = a_0 X^{a_1}.$$

Solving the profit-maximization problem,

$$\max_X \pi = PQ - wX,$$

gives

$$X = \left[\frac{1}{a_1}\frac{w}{P}\right]^{1/(a_1 - 1)}. \tag{5}$$

Impact of Policy Changes on Groundwater Quality

Consider now a policy that sets $P_t = P^*$ for all $t > t^*$. We have the following relationships:

$$\left.\begin{aligned} \partial C_t/\partial X_t &= 0, \text{ for } t - t' < m \\ &= R_{t't}, \text{ for } t - t' > m \end{aligned}\right\} \text{ and } t' > t^*.$$

Hence, the elasticity of C_t with respect to X_t is

$$\epsilon_{tt'} = X_{t'} R_{t't} / C_t. \tag{6}$$

The elasticity of X_t with respect to P_t is, according to the model in equation (5),

$$\eta_t = 1/(a_1 - 1), \text{ for all } t. \tag{7}$$

It follows that the effect of raising P permanently at time t^* by the amount $\Delta P^* = P^* - P_0$ is

$$\Delta C_t/\Delta P^* = \sum_{k=1}^{t} (\Delta X_k^*/\Delta P^*) R_{kt},$$

which in point-elasticity form is, in general,

$$\xi_t = \sum_{k=t^*}^{t} \epsilon_{tk} \eta_k, \tag{8}$$

and using (6) and (7) becomes

$$\xi_t = \sum_{k=t^*}^{t} X_k R_{kt}/C_t(a_1 - 1). \tag{9}$$

These relationships are illustrated in Figure 3 under the assumption that before t^*, $P = P_0$, and input use occurs at fixed time intervals. Under the

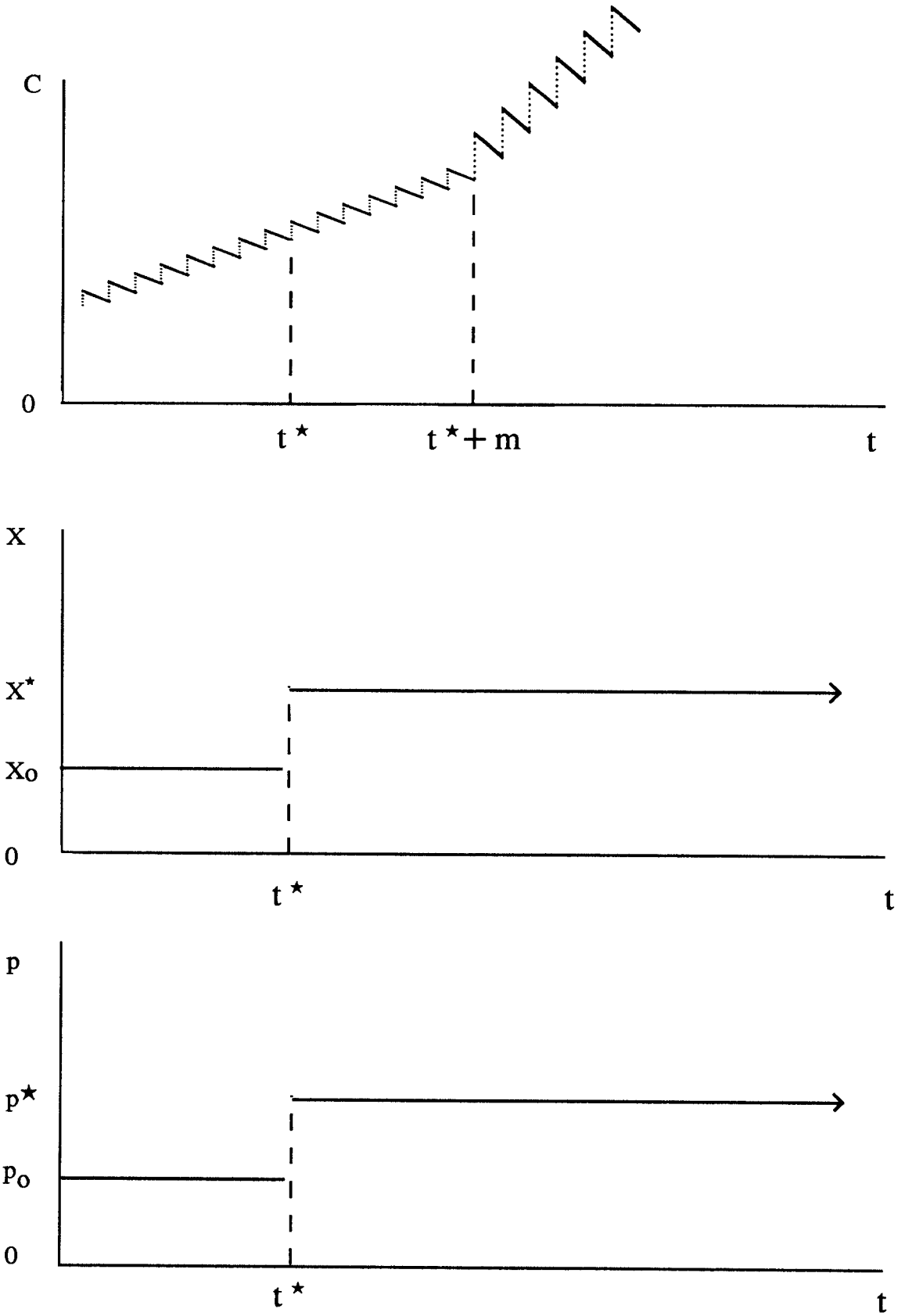


Figure 3. Time Paths of Output Price (p), Input Use (X), and Groundwater Contamination (C) with a Once-and-for-all Change in Price Policy

baseline scenario, input use generates a relatively slow increase in groundwater contamination levels; when policy raises the price of the crop, chemical-use levels increase and the rate of growth in contamination increases. Observe that before t^*, contamination levels increased by the amount ΔC_t each period, whereas after $t^* + m$, contamination levels increase by $\Delta C_t^* > \Delta C_t$ each period (note the delay of m between the time the policy change is implemented and it begins to have an effect on groundwater quality because of the transport time). The elasticity ξ_t measures the percentage increase in C_t for each time period. Note that ξ_t is zero for $t^* < t < t^* + m$ and is an increasing value thereafter.

The analysis of a policy that reduced P once and for all would be similar and would show that a reduction in input-use levels would reduce contamination levels over time. Note, however, that the effect of the policy on groundwater quality would occur with a delay of m.

This simple example illustrates several interesting points. First, equation (8) shows that, in general, the effect of policy on groundwater quality is a function of all of the physical and economic parameters required to obtain ϵ_{tk} and η_k, whether these values are estimated from simple or complex models.

Second, suppose that chemical-input use was sufficiently low such that $C = 0$ for all $t < t^*$ because all of the chemical degrades in the soil during transport ($r = 0$). Then a policy that induced an increase in chemical use would not affect contamination until input use reached the critical level at which r becomes marginally positive. Hence it follows that a policy that increases input use does not necessarily decrease groundwater quality.

More generally, input use will not be at constant intervals and market prices will be changing over time in response to policy and market conditions, and the time path of contamination levels will be much more complicated.

Finally, note that this model applies to a specific site. As discussed in the previous section, it can be assumed that the physical and economic parameters follow well-defined distributions in the watershed. This distribution, in turn, defines a joint distribution in the watershed for C, Q, and X. This joint distribution can be used to represent the watershed statistically as a unit and to conduct policy analysis. For example, it would allow statements to be made about the effect of a policy change on the expected (average) contamination level, or about the probability that contamination at any site in the watershed is less than or equal to a critical value, such as a maximum contamination level set by a risk analysis.

Conclusion

Benefit-cost analysis provides the foundation for developing a framework for integrating the various strands of disciplinary research needed to assess the environmental impacts of agricultural chemical use. The data needed to identify accurately the potential for environmental impacts of chemical use are location-specific and chemical-specific. These information needs include the characteristics of the chemical and the physical environment that provide a basis for estimation of the chemical's mobility and degradation in the environment, and farm-level and field-specific production data that allow the farmer's chemical-use decisions to be modeled.

The heterogeneity of the physical environment means that chemical transport must be modeled at a highly disaggregate level. Thus, farmers' chemical-use decisions must also be modeled at a disaggregate level. Policy issues must be addressed at a more aggregate level, however. The bridge between these two levels of analysis is a statistical representation of the physical environment and the producer population, which provides the basis for statistical aggregation from the highly disaggregate level required for physical models to the more aggregate level of policy analysis. The integration of physical and economic models reveals that, in general, the effect of technological or policy changes on environmental quality will depend on key physical and economic parameters. Considering the demanding data requirements of the integrated physical and economic analysis, a critical issue facing researchers is to identify minimal information sets needed to accurately estimate physical and economic parameters.

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