THE ECONOMIC EVALUATION OF THE SOCIAL COSTS OF AGRICULTURAL GROUNDWATER POLLUTION

by

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APPROVAL

of a thesis submitted by

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ABSTRACT

There is overwhelming evidence that agricultural chemicals make a positive contribution to U.S. agricultural production. In order to determine the net benefit (cost) to society of agricultural chemicals, the social costs and benefits must be quantified and valued. One potential social cost of agricultural chemical use is the human health effects of chemically-contaminated groundwater. In this thesis a multi-disciplinary framework, incorporating physical models and economic production models, is developed to value the health risks of polluted groundwater. This framework can also be used to determine the impacts of agricultural policy on groundwater quality.

In the economic model, farmers jointly make input use, management and land use decisions. Land use decisions determine the environmental characteristics of the land in production. The farmer's economic production model is linked with an environmental damage model (such as a chemical fate and transport model) to determine the amount of groundwater pollution resulting from the use of agricultural chemicals on land with particular environmental characteristics. Toxicology and epidemiology studies are used to estimate the human health risks presented by groundwater contamination, and a contingent valuation method is used to place a value on those risks. The contingent valuation method uses survey techniques to elicit individuals' willingness to pay for a change in the level of groundwater contamination and the accompanying change in health risks.

A case study is presented in order to evaluate the feasibility of linking the chemical fate and transport, economic, and human health models. Although the general physical models needed for this framework are currently available, most of these models are designed to be used by researchers within the respective discipline. Consequently, there are some important gaps in methods and data, including: (1) chemical fate and transport models that do not simulate chemical movement down to the groundwater zone, (2) lack of chemical-specific toxicity and epidemiologic data, and (3) lack of location-specific environmental data. This study illustrates the need for researchers to be aware of the implications and potential applications of their research, both within and outside their fields.

CHAPTER 1

INTRODUCTION

There is overwhelming evidence that agricultural chemicals (fertilizers and pesticides) make a positive contribution to U.S. agricultural production (Headley, 1968; Capalbo and Antle, 1988). There is also evidence that suggests there may be social costs associated with agricultural chemical use due to environmental and human health impacts (Sharp et al., 1986; Berteau and Spath, 1986; Moses, 1987; Coye, 1985). In order to determine if a technology is making a positive net contribution to society, the private and social costs and benefits of that technology must be quantified and valued. The private costs and benefits of agricultural chemical use can be quantified and valued using existing data and methods (Pimentel, 1979), but the social costs and benefits are more difficult to quantify because these impacts involve non-market goods such as environmental amenities and human health. A multi-disciplinary approach is therefore necessary to quantify and value the social costs of agricultural chemical use.

The social costs or externalities of an agricultural production technology can be accounted for if they can be quantified and valued. This process involves: (1) the quantification of the relationship between agricultural production and the environmental sectors it impacts (both directly and indirectly), and (2) the valuation of these

externalities. Figure 1 presents the four major components (and the interaction of these components) that must be considered in the evaluation of an agricultural pollution externality. The interactions among the agricultural production sector, the human health sector, and the agro-ecosystem must be quantified and then valued to determine the extent of these impacts on social welfare. Physical, chemical, and biological models can be used to estimate the impacts of the agricultural production technology on the agro-ecosystem and on human health.

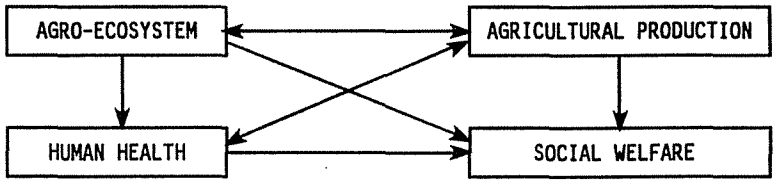


Figure 1. Relationships among the major components in the evaluation of agricultural pollution externalities.

As with an evaluation of any agricultural pollution externality, the evaluation of groundwater pollution resulting from agricultural chemical use would include an analysis of the impacts on the four sectors presented in Figure 1. The characteristics of the agro-ecosystem and the producer's management decisions jointly determine the amount of damage to the agro-ecosystem (e.g., groundwater contamination), which in turn can impact human health. To quantify the damage to the agro-ecosystem, it is first necessary to model the movement of the chemical within the agro-ecosystem. There is a variety of chemical fate and transport models that can be used to trace chemical movement. Once the distribution of the chemical within the agro-ecosystem is

established, one can begin to evaluate the impacts of the chemical. The evaluation of health risks requires information on the levels and duration of exposure to the chemical and the types and extent of health risks presented by that exposure.

To conduct an economic evaluation of the net social benefits of agricultural chemicals, benefit-cost methodology can be used. Benefit-cost analysis provides a useful framework for the evaluation of environmental quality problems (Freeman, 1979; Lave and Seskin, 1977). It is based on neoclassical welfare theory; i.e., an individual's preferences are accepted as the basis for measuring welfare gains and losses associated with alternative pesticide use. For a benefit-cost analysis, the private and social costs and benefits of the technology must be identified, quantified, and valued. Benefit-cost analysis of agricultural chemical technology consists of the following steps. The first is to specify the economic production model. This step provides the information that can be used to calculate private benefits; it will also determine the producer's use of chemical inputs, which is needed to estimate the amount of potential groundwater pollution. The second step is to use the physical and biological models to describe the dynamics of the agro-ecosystem. In the evaluation of groundwater pollution, this step involves tracing the movement of the chemical through the soil profile, estimating the extent of groundwater pollution, and determining the physical and biological effects of the groundwater pollution on the agro-ecosystem.

The third step is to use a risk assessment model to quantify the human health risks of polluted groundwater. There are numerous ways to

assess health risks and to measure and estimate exposure. The particular risk assessment model used in an analysis will depend on factors such as the chemical in the groundwater, the types of health hazards the chemical presents, and the type and length of exposure.

The fourth step is to value the private and social costs and benefits quantified in the previous three steps. The valuation will require the use of market information (such as input and output prices) to value private costs and benefits and non-market techniques (hedonic pricing, contingent valuations, etc.) to value public costs and benefits such as health effects.

The fifth step of the economic evaluation framework is to make a determination of net benefits (costs). The private and social costs and benefits are summed for each period and discounted to determine the net present value of the stream of costs and benefits; if the net present value is positive (negative), the use of agricultural chemicals results in positive benefits (costs) to society. If all the social and private costs and benefits were not valued in the fourth step, the total net present value of the costs and benefits cannot be determined.

Determining the total cost to society of a technology is important for policy analysis purposes. If a technology creates a net cost to society, the magnitude of the social costs can be a factor in the determination of society's optimal policy regarding agricultural chemicals. For example, two possible regulatory options concerning chemicals are a total ban on use of the chemical, or restrictions on its use. Banning the chemical would be optimal if total costs exceed total benefits for all quantities of chemical application. Restricting chemical use may be

optimal if total benefits exceed total costs over some range of chemical use.

This thesis develops a methodology for evaluating the social costs of agricultural chemical pollution of groundwater and consists of three major components:

(1) The development of a framework for evaluating agricultural chemical pollution of groundwater. This framework consists of multidisciplinary components needed to quantify and value social costs for a benefit-cost analysis.

(2) The linking of the physical and biological models, human health hazard models, and economic models involved in (1) above.

(3) A case study on the use of the herbicide Ro-Neet in sugar beet production in the Yellowstone Valley of Montana. An assessment of available methods, models, and data bases is presented and will indicate the feasibility of using this framework given the current state of knowledge.

The outline of this thesis is as follows. Chapter 2 presents the necessary considerations for modeling the agricultural production process with explicit consideration given to incorporating the use of agricultural chemicals. Chapter 3 discusses chemical fate and transport models. Chapter 4 presents a method of linking an economic production model and a chemical fate and transport model. Chapter 5 discusses techniques of valuing human health risks. Chapter 6 discusses current methods of estimating human exposure to agricultural chemicals and health risks of exposure. Chapter 7 provides an overview of data bases

that could be used in an application of the framework discussed in Chapter 4. Chapter 8 is a case study that illustrates the steps and considerations for a valuation of the social costs of groundwater pollution in the Yellowstone Valley of Montana. Chapter 9 presents some concluding remarks and discusses research recommendations.

CHAPTER 2

THE ECONOMIC PRODUCTION MODEL

The neoclassical theory of production is commonly used to analyze a firm's production decisions. Neoclassical theory is centered on the assumption that a firm's short-run objective is to maximize profits under conditions of costless information and certainty; maximizing decisions are assumed to be made at the beginning of the production process.

The Neoclassical Production Model

The neoclassical model is a one period model in which prices and output are known with certainty. The firm's objective function can be expressed as

$$\max_{X} J(x,y,\gamma), \text{ where } J(x,y,\gamma) = pq - wx$$

$$q = f(x,y,\theta)$$

where J is returns to fixed factors, p is output price, q is output, w is the vector of variable factor prices, $x = (x_1, \ldots, x_n)$ is the vector of variable input factors, y is the vector of fixed factors, θ is a vector of parameters of the production function, and γ is a vector of input and output prices and production function parameters, $\gamma = (p,w,\theta)$. The first order condition is:

$$\partial J/\partial x_t = p\partial q/\partial x_t = w_t, \quad t = 1, \ldots, n \qquad [2.1]$$

The first order conditions equate the value of the marginal product (VMP) of input x_t to the marginal factor cost (MFC) of x_t, which in this model is equal to the price w_t to x_t. Solving the set of first order equations results in variable factor demands that are functions of prices, level of q, fixed factors, and parameters of the production function:

$$x^* = x^*(p,w,y,\theta) \qquad [2.2]$$

where the * denotes the optimizing input level.

This static and deterministic model is not always the most realistic representation of a production process or of profit maximization, and so the relaxation of some of these assumptions may result in a more representative model. Two modifications to the static production model which are important for the research in this thesis will be discussed: incorporating dynamics and incorporating risk.

Dynamics of Agricultural Production

In a static model all input decisions are made at the beginning of the production process, whereas in a dynamic or sequential decision model production decisions are made over time. In the case of agricultural production, input decisions are often made throughout the production process because of the time dimension of biological processes. For example, a pest infestation that was not existent or apparent at the beginning of the process may become apparent at some point in the production season; the producer is then faced with the intra-seasonal decision of whether or not to apply a pesticide. In this example the producer is using information gathered over time to maximize profits. Because time is a factor in this producer's production decisions, a

dynamic production model is a better representation of this production process than a static model.

Antle (1983b) presents the two general forms of dynamics that can be incorporated into a production model: input dynamics and output dynamics. A production process may exhibit both input and output dynamics. Input dynamics are characterized by output that is a function of a sequence of inputs:

$$q_t = f_t(x_t, x_{t-1}, \ldots, u_t)$$

where q_t is output at time t, x_t is the input vector, and u_t is the random production disturbance in time t. Output dynamics involve a series of decisions made over time and are characterized by a final output that is dependent on a sequence of intermediate outputs:

$$q_t = f(x_t, q_{t-1}, q_{t-2}, \ldots, u_t).$$

The production model can be formulated as a multistage or multiperiod decision problem. A multistage problem divides a production process into a sequence of production stages, whereas a multiperiod problem examines a sequence of temporally related production processes. For example, the production of a crop that reaches maturity in one season may exhibit output dynamics and the producer may have a multistage decision problem; the crop will produce a series of intermediate outputs during that season and because of these observable intermediate outputs, the producer may be faced with intra-seasonal decisions. The objective function of the multistage model (having T stages) is of the form:

$$\max E(p_T q_T - \sum_{t=1}^{T} g_t w_t x_t)$$

where $E(\bullet)$ is the expectation operator, q_T is the final output, and g_t is a discount factor.

With a multiperiod production process, the producer may be faced with inter-seasonal decisions such as the type of crop rotation to implement. The multiperiod model objective function is of the form

$$\max E\left(\sum_{t=1}^{T} J_t g_t\right)$$

where $J = p_t q_t - w_t x_t$

In both multiperiod and multistage dynamics, the producer's objective function is to maximize the present value of producer surplus (return to fixed factors).

Solutions to these dynamic optimization problems can be obtained using the dynamic programming algorithm. This algorithm relies on the segmentation of a problem into a sequence of smaller subproblems, called stages, with each stage requiring some type of decision. Each stage has a number of states associated with it; a state describes the condition of the process at a particular stage. The decision made at a stage will determine the transformation of the current state into a state associated with the next stage. The solution to a dynamic programming problem consists of the sequence of optimal decision rules for each stage (Bellman, 1957).

The dynamic programming algorithm starts at the end of the planning horizon and works backwards to the beginning by solving a series of recursive equations. The following example illustrates the form of these equations:

Stage

T $\quad \max_{x_T} J(x_T) \Rightarrow x_T^*$

T-1 $\quad \max_{x_{T-1}} J[x_{T-1}, x_T^*(x_{T-1})]$

t $\quad \max_{x_t} J[x_t, x_{t+1}(x_t), \ldots, x_T^*(x_{T-1})]$

.
.
.

1 $\quad \max_{x_1} J[x_1, x_2^*(x_1), \ldots, x_t^*(x_{t-1}), \ldots, x_T^*(x_{T-1})]$

where x_t^* is the decision rule for stage t and T is the number of stages. The optimal decision rule at stage t is conditional on decisions to be made in stages 1, ..., t-1 and on optimal decision rules for stages t+1, ..., T. (See Burt and Allison, 1963, for an application of the dynamic programming algorithm to agricultural production.)

In a dynamic, stochastic model with a risk-neutral producer, the producer's objective is to maximize the present value of profit by choosing the optimal input sequence. In the short-run, with fixed costs and the state of technology held constant, maximizing profit is equivalent to maximizing producer surplus. This dynamic problem can be approached with the objective function for each stage being expressed as:

$$\max_{x_t} E\,(J|I_t)$$

where I_t is the information set in stage t, and $I_t = (I_0, \ldots, I_{t-1}, q_{t-1}, w_t, x_{t+1}^*, \ldots, x_T^*)$, I_0 is the initial information set, and $I_0 = (w_1, \ldots, w_T, v_0)$, and v_0 is the vector of parameters of the producer's initial expectations of future prices and output.

The first order conditions of the dynamic model differ from those of the static model. To illustrate the differences, consider a two-stage dynamic model:

$$q_1 = f_1(x_1, u_1)$$

$$q_2 = f_2(x_2, q_1, u_2)$$

where q_t is output at the end of stage t, x_t is the variable input at stage t, and u_t is the random production disturbance occurring during stage t.

In stage 2 the producer's maximization can be expressed as:

$$\max E[(pq_2 - w_1x_1 - w_2x_2)|v_2]$$

where p is the price of q_2, w_t is the price of input x_t, and v_t is the vector of parameters of the producer's expectations at time t of future prices and output.

The first stage input, x_1, is chosen based on the producer's expectations of future prices and output, and decision rules for future optimal inputs, in this case x_2^*. In stage 1 the producer's maximization can be expressed as:

$$\max E[(pq_2 - w_1x_1 - w_2x_2^*)|v_1]$$

The first order condition for stage 1 is:

$$E_1\left[p\,\frac{\partial q_2}{\partial x_1} + \frac{\partial q_2}{\partial x_1} \cdot \frac{\partial x_2^*}{\partial x_1}\right] = E_1\left[w_1 + w_2\,\frac{\partial x_2^*}{\partial x_1}\right] \qquad [2.3]$$

Recall from equation [2.1] that the first order conditions of the neoclassical model equate the VMP at t to the w_t, and so factor demands are a function of all variable factor costs, output price, and fixed factors at time t. The first order condition for this dynamic model

equates the expected value of marginal product (EVMP) to the expected marginal factor cost (EMFC). In the dynamic case, $EVMP_t$ (the left-hand-side of equation [2.3]) equals the direct effect of x on output plus the indirect effect of x on output due to its effect on input requirements in future stages, and EMFC (the right-hand-side of equation [2.3]) equals the factor cost for that stage plus the effect on cost in the future stages. The production decisions in the dynamic model are functions of previous input decisions, and the effect of current decisions on future decisions (Antle, 1988). Therefore, because the producer updates his information set, factor demands are functions of output over time

$$x_t^* = x_t^*(w_t, w_{t+1}, \ldots, w_T, q_{t-1}, q_{t-2}, \ldots, I_t) \qquad [2.4]$$

Imposing any type of restrictions on the production model has certain implications about the behavior of the firm (producer). A dynamic production model implies the producer makes decisions over time and can use newly-acquired information to optimize. Imposing a dynamic structure on a production model also has implications regarding the separability of inputs. A production function exhibiting output dynamics can be expressed as

$$q_t = f[x_t, q_{t-1}, u_t]$$

$$= f[x_t, f_{t-1}(x_{t-1}, q_{t-2}, u_{t-1}), u_t]$$

where x_t is a vector of inputs and u_t is a random production disturbance.

Because of the time factor in this example, the x_{t-1} inputs are separable from the x_t inputs, but the x_t inputs are not separable from the x_{t-1} inputs. Being able to assume separability of inputs can simplify the task of econometric estimation.

Risk in Agricultural Production

Because agricultural production is not deterministic, it may be necessary to incorporate risk into the production model. The agricultural production process is dynamic and production is not instantaneous so that output and all prices usually are not known with certainty until after the input decisions have been made. There are two general types of risk that apply to agricultural production: production risks, such as pests, weather, etc., and price risks due to changes in market conditions.

Incorporating production risk into a production model can be accomplished by interpreting output as a random variable. The firm's revenue and profit are stochastic if output is a random variable. When profit is stochastic, the producer's objective function is often assumed to be to maximize expected utility of profit. In the case of risk neutrality, the marginal utility of profit is constant; utility is a monotonic transformation of profit and so maximizing expected utility of profit is equivalent to maximizing expected profit. In a static model where the producer is risk neutral and a price taker, his objective function is

$$\max_{x} EU(J) = E(J) = E(pq - wx) \qquad [2.5]$$

$$= E(p)E(q) + COV(p,q) - wx \qquad [2.6]$$

where $E(\bullet)$ is the expectation operator, J is profit, U(J) is the utility of J, and COV(p,q) is the covariance between p and q (Newberry and Stiglitz, 1981). Assuming that price and output are statistically independent, [COV(p,q) = 0], the first order condition is

$$\partial E(J)/\partial x = 0$$

which implies

$$E(p)\partial E(q)/\partial x = w \qquad [2.7]$$

The left-hand-side of equation [2.7] is the expected value of the marginal product (EVMP) which is equated to the marginal factor cost (MFC), or factor price w.

The utility function of a risk-averse producer exhibits diminishing marginal utility. Utility of profit is not a monotonic transformation of profit, and so maximizing expected utility of profit is not equivalent to maximizing expected profit. The risk-averse producer's objective function is:

$$\max_x EU(J) = EU(pq - wx)$$

The first order condition can be expressed as:

$$E(p)\partial E(q)/\partial x + E(p)\mathrm{cov}(U', \partial q/\partial x)/E(U') - w = 0 \qquad [2.8]$$

where U' is marginal utility of income (Antle, 1988).

Equation [2.8] can also be interpreted with the use of the certainty equivalent (CE). The CE is the dollar amount that the firm could receive with certainty if it were insured against risk. CE is a measure of producer welfare.

$$CE = E(J) - R$$

where R is a risk premium (see Newberry and Stiglitz, 1981, and Pearce and Nash, 1981).

In this model, maximizing profit (J) is equivalent to maximizing producer welfare (producer surplus):

$$\partial EU(J)/\partial x = \partial CE/\partial x = \partial E(pq-wx)/\partial x - \partial R/\partial x = 0 \quad [2.9]$$

$$= E(p)\partial E(q)/\partial x - \partial R/\partial x - w = 0 \quad [2.10]$$

where $\partial R/\partial x$ is the marginal risk premium. Because equation [2.8] is equivalent to equation [2.10], the second term on the right-hand-side of both equations can be interpreted as the marginal risk premium. In a static risk-neutral model, $R = 0$. If in the risk-averse model $\partial R/\partial x$ is not equal to zero, then the producer's risk attitude affects his optimizing decisions. If the marginal risk premium is equal to zero, the first order condition of the risk-averse case is equal to that of the risk-neutral case, and the factor demands in both cases will be equal. If the covariance is not equal to zero, then the marginal risk premium is the amount by which the risk-averse producer's input use differs from that of the risk-neutral producer. A negative covariance (positive marginal risk premium) indicates that the risk-averse producer will use less of the input than the risk-neutral producer and the input is said to be marginally risk-increasing; this negative covariance has an equivalent effect on input usage as an increase in MFC. If the covariance is positive (negative marginal risk premium), the risk-averse producer will use more of the input than the risk-neutral producer and the input is said to be marginally risk-reducing. This positive covariance has an equivalent effect on input usage as does a decrease in MFC.

As Antle (1983b) demonstrates, in some cases it may not be necessary to incorporate risk into the production model. However, in models that involve dynamics or a risk-averse producer, it is important to consider risk. In a dynamic model, incorporating risk is important whether the producer is risk-neutral or risk-averse because in both cases risk

affects resource allocation and expected productivity. As stated previously, a dynamic model may be a multistage model or a multiperiod model. Although these models have different objective functions, the solutions of the first order conditions result in factor demands of the form

$$x_t^* = x_t^*(x_{t-1}, x_{t-2}, \ldots, q_{t-1}, q_{t-2}, \ldots, w_t, u_t)$$

where u_t is the parameter vector defining the joint probability distribution of current and future output and future prices. In both the multistage and multiperiod models the factor demand is a function of the distribution of current and future output and future prices, indicating that optimal resource allocation is dependent on price and output risk.

When a production function is stochastic, the producer's utility function can be expressed as a function of profit and risk attitudes:

$$U_j = U(J_j, g_j)$$

where U_j is the j^{th} producer's utility function, J_j is profit (and is stochastic), and g_j is a vector of parameters that measures the j^{th} producer's risk attitudes. Therefore, the producer's production decisions (determined by maximizing expected utility), are affected by the stochastic profit function, J_j, and his attitudes towards risk, g_j.

Determining the risk attitudes of every individual in a population may be impossible due to large data requirements, and to assume that producers' risk attitudes are homogeneous may be an inappropriate simplification. Antle (1987) presents a method of estimating the distribution of producers' risk attitudes, assuming producers use similar production technology. A moment-based approximation of the profit distribution allows for the estimation of the derivatives of the expected

utility function; the sign of the derivatives of the utility function can then be inferred from the derivatives of the expected utility function.

In summary, the neoclassical model is a static and nonstochastic model in which producer behavior is determined by prices and the production function. However, agricultural production can involve risk and decision-making over time, and so the neoclassical production model may have to be modified to include these effects on producer behavior. Incorporating risk into the production model results in producer behavior that is a function of the producer's risk attitudes. Incorporating dynamics into the production model results in producer behavior that is a function of time.

CHAPTER 3

CHEMICAL FATE AND TRANSPORT MODELS

Mathematical models of chemical fate and transport are tools with which to evaluate nonpoint source contamination of groundwater. Chemical fate and transport models simulate the movement of a chemical in the environment. In the case of water quality, the output of these models may be the effects of chemical use on surface water quality, groundwater quality, or both. Concern over groundwater contamination is a relatively recent development (relative to concern over surface water contamination), and models that predict chemical leaching to groundwater are less developed than those that predict chemical runoff to surface water (Mulkey et al., 1986).

Chemical loading models can be divided into three broad categories: research models, screening models, and management models (Wagenet and Hutson, 1986). Research models provide quantitative estimates of water and solute movement and have extensive data requirements. Screening models can be used to evaluate the potential environmental effects of a policy that is implemented over a broad geographical area. They are designed to be simple, quick, relatively inexpensive to use, and supported by available data bases (Bailey and Swank, 1983). Management models are less data-intensive and less quantitative, and consequently the ability of a management model to predict water and solute movement

is less precise than that of a research model. According to Wagenet and Hutson (1986, p. 316), "In the case of all management models, the ability of the model to quantitatively estimate pesticide fate under field conditions is questionable." Managerial models are usually designed to incorporate changes in management decisions (such as tillage practices and crop rotations). Many managerial models are field-scale models because that is the area appropriate to agricultural decision making.

Pollution of the agro-ecosystem due to agricultural chemicals is dependent on hydrologic, ecologic, agronomic, social, and economic subsystems. Figure 2 illustrates the factors that affect the behavior of agricultural chemicals in an agricultural watershed. Natural inputs (e.g., climatic characteristics) and management inputs (e.g., chemical use, land use, and cultural or tillage practices) combine with the environmental characteristics of the water shed to determine the fate and transport of the agricultural chemical. Figure 2 indicates that chemicals can be transported via air and water. The focus of this thesis is the transport of dissolved chemicals to groundwater (indicated as the subsurface flow of Figure 2).

To model potential loading to groundwater, a chemical transport 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. Figure 3 illustrates the zones involved. The zone of saturation is the area in which all the void spaces are filled with water; in the aerated (or unsaturated) zone, the void spaces are filled with both air and water. The soil water zone lies below the ground

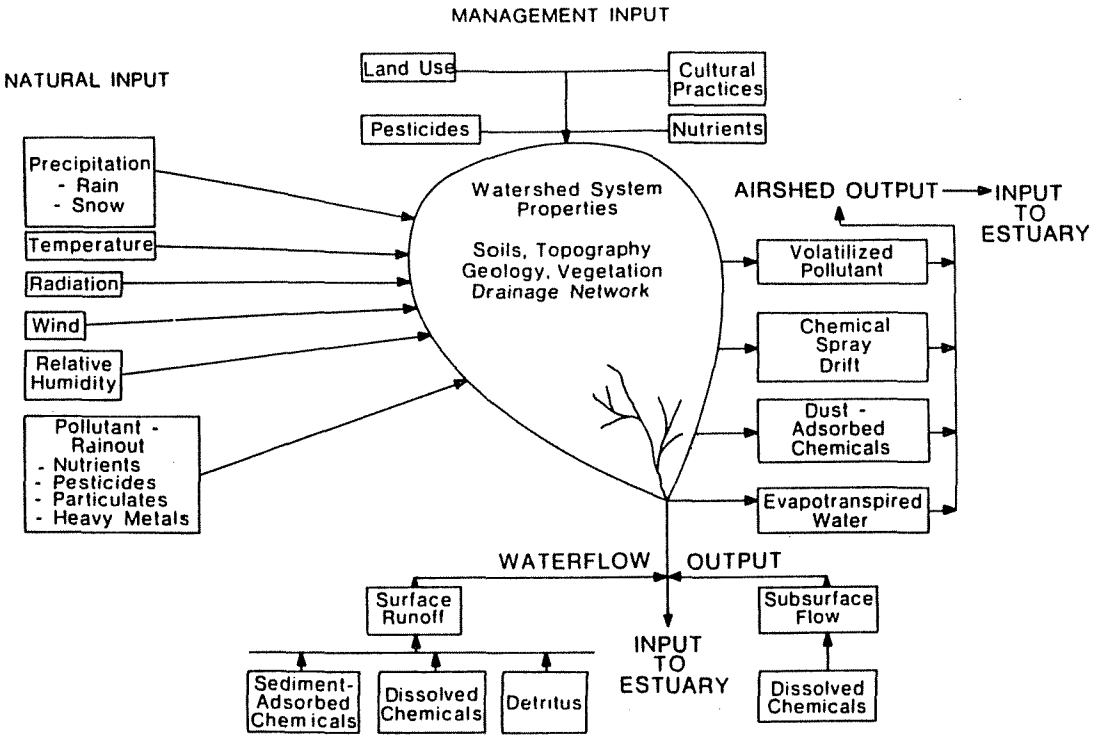


Figure 2. Factors influencing the behavior and export of agricultural chemicals from an agricultural watershed (Bailey and Swank, 1983, p. 30).

surface and extends down through the root zone; hence this zone is also called the root zone. The intermediate or vadose zone lies below the root zone and above the groundwater zone (Bear and Verruijt, 1987). Some models simulate chemical transport only part way through the unsaturated zone (e.g., through the root zone). If the unsaturated zone model does not simulate chemical transport down to the unsaturated zone, the model may not give a reasonable estimate of groundwater loading. In order to estimate the potential groundwater concentration of a chemical, it would be necessary to link the chemical transport model with a groundwater model. Some loading models may have this linkage built in, and some require the user to make this linkage.

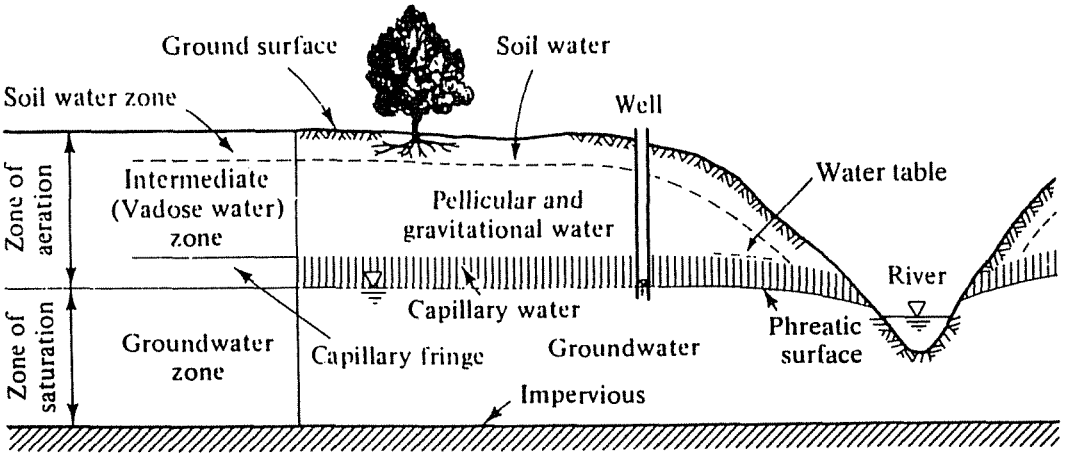


Figure 3. Subsurface moisture zones (Bear and Verruijt, 1987, p. 4).

Before examining the chemical transport models, it is helpful to have an understanding of the behavior of chemicals in soil and water. Rao et al. (1983) provide an overview of the behavior of pesticides in soil and water; much of the following information is taken from their work.

The fate of a pesticide applied to soil depends largely on two of the pesticide's properties: persistence and solubility. Persistence is a measure of a chemical's rate of degradation and is usually measured in terms of the chemical's half-life.

Solubility and sorption determine how a compound partitions among soil, water, and air phases and affect whether the chemical is moved primarily with sediment or water. According to Rao et al. (1983, p. 2), "[Solubility is] probably the single most important property influencing a pesticide's movement with 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 may move down in the soil with the soil water. Desorption, the detachment of pesticide molecules from soil particles, may also occur as water moves through the soil (e.g., rain or irrigation). An inverse relationship exists between the solubility of a pesticide and its sorption to soil. A partition coefficient value (K_p) is used to describe the ratio of pesticide concentration in the adsorbed phase and the solution phase; the K_p is determined by the chemical's properties such as solubility and melting point. The K_p can be expressed as a ratio of the chemical concentration in the soil or solid phase (C_s), and the concentration in solution (C_w):

$$K_p = C_s/C_w.$$

The smaller the K_p, the greater the concentration of pesticide in solution. The greatest threat to groundwater through leaching is a pesticide with a small K_p and a long half-life. However, a pesticide with a small K_p and a short half-life may also pose a threat of leaching into groundwater if heavy precipitation closely follows application.

The transport of a chemical from the application site down through the unsaturated zone is dependent on hydrologic, soil, and chemical characteristics. In many chemical transport models these processes are incorporated into hydrologic, soil, and chemical components. Although the structure of a model may vary, most models contain some standard components. Novotny (1986) describes the basic components that are included in a nonpoint pollution model. The basic components needed to estimate potential groundwater loading from agricultural chemicals include:

(1) Surface Runoff Generation Component: Describes the transformation of precipitation into runoff. (Precipitation will become runoff or infiltration.) Surface runoff is a function of surface storage, evapotranspiration (the sum of plant and soil evaporation), and snow accumulation and melt. The USDA 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. This SCSCN model uses the equation

$$Q = \frac{(P - 0.2S)^2}{(P + 0.8S)}$$

where Q is daily runoff, P is daily rainfall, and S is a retention parameter. The retention parameter is expressed as a function of the curve number (CN) (Williams and LaSeur, 1976):

$$S = \frac{1000}{CN} - 10$$

(2) Soil and Groundwater Component: Describes chemical movement through the unsaturated soil zone and may also describe movement into the saturated zone. (As previously mentioned, not all models trace the movement all the way 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 chemicals. A chemical 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 USDA/Soil Conservation Service (1978) summarizes the USLE:

$$A = RxKxLSxCxP$$

where A is the computed soil loss in tons per acre per year, R is the rainfall factor, K is the soil erodibility factor, LS is the slope length ratio factor and slope gradient ratio factor, C is the cover-management (cropping-management) factor, and P is the erosion control practice factor.

(4) Soil Adsorption and Desorption Component: Estimates the partitioning of a chemical between adsorbed particles and dissolved chemical. This is the K_p mentioned earlier. 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.

Of the chemical fate and transport models currently used, many are non-steady-state equilibrium, continuous simulation models. An equilibrium model assumes the physical and biological processes rapidly achieve a state of equilibrium, and non-steady-state (or dynamic) models are designed so the ecosystem varies with time due to changing conditions or due to changing inputs. Continuous simulation models are usually based on a series of differential equations and therefore model changes in chemical concentration in the soil profile over time. Long-term simulations are used in chemical fate and transport models because simulating potential chemical loading for one year may not be indicative

of typical chemical loading since potential loading is dependent on rainfall patterns. Long-term simulations are better able to consider the typical variation in rainfall pattern as well as the impact of a crop rotation on potential chemical loading. The results of long-term simulations can be used to determine long-term average values (e.g., long-term average potential loading), or frequency distributions (e.g., of annual potential groundwater loading).

Mulkey et al. (1986) have defined a system of general mass-balance equations that depict the changes in the chemical concentration profile over time. The following mathematical model is taken from that research:

$$\sum_{i=1}^{n} \frac{\partial (C_i V_i)}{\partial t} = \sum_{i=1}^{n} \frac{\partial}{\partial x} D_i \frac{\partial (C_i V_i)}{\partial x} - \sum_{i=1}^{n} \frac{\partial (C_i V_i \nu_i)}{\partial x} + \sum_{i=1}^{n} APP_i(t) - \sum_{i=1}^{n} \sum_{j=1}^{m} k_j C_i V_i \qquad [3.1]$$

where

C_i = concentration of the pesticide in the i^{th} environmental compartment, M/L^3

V_i = volume of the i^{th} environmental compartment, L^3

D_i = coefficient of dispersion (includes diffusion), L^2/T

ν_i = velocity vector for the media of the i^{th} compartment, L/T

$APP_i(t)$ = pesticide application rate for the i^{th} compartment on day t, M/T

k_j = pesticide transformation rate for the j^{th} process, $1/T$

n = number of environmental compartments or media

m = number of transformation processes acting on pesticide within each compartment or media, assuming first order kinetics

x = space dimension, L

t = time, T

where L is a measure of space, T is a measure of time, and M is a measure of mass.

The left-hand-side term of equation [3.1] represents the change over time of the amount of chemical in a soil compartment. The first term in the right-hand side of this equation represents all of the chemical dispersion processes; dispersion is the mixing or spreading of solute caused primarily by variations in velocity (Carsel et al., 1988b). Diffusion, one of the dispersion processes, is the spontaneous movement of a substance in solution or suspension; this movement is not a function of water flow. The second term represents the advection processes, the movement of the chemical with water. If surface runoff does occur, this advection term will model the transformation of precipitation into runoff and infiltration. (Both dissolved and adsorbed chemical can move with runoff water.) The third term represents the application of the chemical, and the fourth term represents all of the transformation processes acting on the chemical, including degradation.

The management-oriented chemical transport models developed in recent years include the Groundwater Loading Effects of Agricultural Management Systems (GLEAMS), the Pesticide Root Zone Model (PRZM), and the Hydrological Simulation Program-FORTRAN (HSPF). These models have several shortcomings. One major disadvantage is that they cannot be used to model the movement of chemicals down to deep groundwater. For

example, GLEAMS models movement down through the root zone, but not through the vadose zone. HSPF models movement down to shallow groundwater, but not to deep groundwater. PRZM models movement within and below the root zone and can simulate groundwater loading only if the depth to groundwater is sufficiently shallow. Although all three models can simulate the "potential loading" of a contaminant, this estimate is not always a meaningful estimate of the actual loading to groundwater. If there is a significant distance between the lowest soil zone modeled and the saturated zone, then at best the model will provide an upper bound on potential groundwater loading. Therefore, applying these models to get an estimate of potential chemical loading may be suitable in two circumstances: (1) if the depth to groundwater is sufficiently shallow that the pollutant loading estimated by the model feeds directly into the saturated zone, and (2) if shallow groundwater moves laterally within the unsaturated zone and then feeds into another water body, e.g., surface water. Research is currently underway to develop a vadose model that could be linked with a root zone model to simulate chemical movement down to the saturated zone. Another shortcoming of these three models is that none of them accurately predict levels of concentration; rather, they are used to compare the potential water loading effects of different management scenarios. That is, a simulation for a specific site can be run and then certain model parameters can be adjusted to reflect changes in management practices. Then a second simulation can be run and the results of the two simulations can be used to compare the relative potential loadings of the two management scenarios.

PRZM, GLEAMS, and HSPF all follow the same general steps in estimating chemical movement through the unsaturated zone. To provide a clearer view of some of the processes involved, all three models will be briefly discussed, with a more detailed description of PRZM provided in the appendix. To give the reader an idea of the input and output of these models, model applications are also presented.

Management-Oriented Chemical Transport Models

The Pesticide Root Zone Model (PRZM)

Carsel et al. (1984) provide a detailed description of the Pesticide Root Zone Model. PRZM was designed by the Environmental Protection Agency to evaluate pesticide leaching threats to groundwater under differing agricultural crops and management practices and was developed to be used in pesticide registration. PRZM does not model nutrient movement. It is a dynamic, compartmental model for use in simulating pesticide movement in the unsaturated soil systems within and below the root zone. PRZM is divided into two main components: hydrology and chemical transport. The chemical fate and transport mechanisms represented include advection, dispersion, sorption, degradation, and plant uptake. Water balance equations are used to determine runoff, evapotranspiration, and infiltration. The PRZM soil profile is divided into as many soil layers or compartments as necessary to adequately represent the profile. For each compartment the model solves a finite difference advection-dispersion equation on a daily basis.

In the PRZM model three basic components must be solved to estimate potential pesticide loading threats to groundwater: (1) water balance

in the soil profile, (2) erosion from the soil surface, and (3) chemical transport in the soil. These three components contain the four elements of groundwater contamination models described earlier (Novotny, 1986). PRZM's water balance component includes surface runoff generation (Novotny's first component) as well as the routing of water through the root zone. The erosion component in PRZM estimates soil loss due to erosion (Novotny's third component). PRZM's chemical transport component requires the input of the chemical's partition coefficient (K_p) and degradation rate (Novotny's fourth component) and simulates chemical movement through the root zone (Novotny's second component). The three PRZM components are linked together in the advection-dispersion equations. A description of these equations is given in the appendix.

The Groundwater Loading Effects of Agricultural Management Systems (GLEAMS) Model

Groundwater Loading Effects of Agricultural Management Systems (GLEAMS) is a mathematical model developed for field-size areas to evaluate the effects of agricultural management systems on the movement of agricultural chemicals within and through the root zone. GLEAMS, an extension of Chemicals, Runoff, and Erosion from Agricultural Management Systems (CREAMS) (Knisel, 1980), was developed to utilize the management-oriented, physically-based CREAMS model and considers the vertical movement of agricultural chemicals. GLEAMS contains four basic components: (1) hydrology component, (2) erosion component, (3) pesticide component, and (4) nutrient component.

The Hydrological Simulation Program-FORTRAN (HSPF) Model

Barnwell and Johanson (1981) provide a brief description of the HSPF model. A more comprehensive description can be found in Johanson et al. (1984). HSPF, a basin-scale model, was developed for the Environmental Protection Agency and is a descendant of the Stanford Watershed Model. It is a comprehensive program for modeling runoff, sediment, pesticides, nutrients, and other water quality constituents from urban, agricultural, and other land uses. HSPF is a data-intensive model and requires large amounts of computer time for simulation. HSPF evaluates Best Management Practices (BMP) effects on runoff and pollutant loadings into field sized areas, and then links these runoff and pollutant loadings with models to assess the quality in receiving waters resulting from BMPs used on the watershed. HSPF does model chemical transport through soil, but it was designed primarily to evaluate surface water quality, not groundwater quality. Chemical loading of shallow groundwater is included in this model because shallow groundwater can transport contaminants laterally to surface water. In HSPF, shallow groundwater is modeled as a medium of transport for the chemical, not as a receiving body of water; consequently, deep groundwater is not modeled. HSPF could be used to model secondary effects of shallow groundwater contamination such as surface water contamination. Also, HSPF is of interest because it includes a risk assessment component.

HSPF consists of three application modules: PERLND, IMPLND, and RCHRES. PERLND simulates a pervious land segment with homogenous hydrologic and climatic characteristics. IMPLND simulates impervious

land segments with little or no water infiltration. RCHRES simulates the processes occurring in a single reach of an open channel or a completely mixed lake. Water movement is modeled along three flow paths: overland flow, interflow, and groundwater flow. To estimate the water quality effects of agricultural chemicals on a basin stream or river, the amount of loading from all three waterflow paths is summed. Because HSPF can model a larger variety of situations than the other models discussed, it is more complicated than GLEAMS and PRZM. However, HSPF models the groundwater loading processes in much the same way as GLEAMS and PRZM.

The Chemical Migration and Risk Assessment Methodology (CMRA) developed by Battelle Pacific Northwest Laboratories has been incorporated into HSPF. The methodology includes submodels of sediment transport by particle size, pesticide routing and degradation kinetics, and sediment-pesticide interaction. A frequency analysis provides a statistical summarization of time-varying contaminant concentrations and provides the link between simulated instream toxicant concentrations and risk assessment.

Model Applications

Pesticide Root Zone Model (PRZM) Application

The following information describing the PRZM application is derived from the work of Donigian and Carsel (1987), "Modeling the Impact of Conservation Tillage Practices on Pesticide Concentrations in Ground and Surface Waters." This PRZM application illustrates the linking of an unsaturated zone model with a saturated zone model. PRZM can model

pesticide movement down to the saturated zone, providing the depth to the saturated zone is relatively shallow. To determine probable concentrations of a pesticide in groundwater, the Analytical Transient One-, Two-, and Three-Dimensional Simulation of Water Transport in the Aquifer System (AT123D) is linked with PRZM. This application evaluates different pesticides and different tillage practices applied to a corn and soybean rotation in the Lake Erie Basin, and determines the probable concentrations of chemicals in groundwater and surface water under different tillage practices.

In order to incorporate the effects of the tillage practices into the PRZM model, model parameters are adjusted to reflect these effects. The parameters selected to represent these impacts on soil hydrology, sediment erosion, and chemical processes are: the SCSCN, the soil field capacity, the soil organic carbon content, and the USLE. To estimate groundwater loading, the chemical transformation rate is needed. Donigian and Carsel point out that available data on transformation rates of pesticides in subsurface soil horizons and groundwater are limited, and that the transformation rate will depend on soil conditions (pH, temperature, etc.). Based on limited laboratory data, the authors assume that the lower horizon decay rates of the pesticides examined are 50 percent of the rate in the surface horizon.

In this situation, the depth to groundwater is sufficiently shallow that PRZM can model chemical movement all the way through the unsaturated zone, and the chemical loading simulated can be fed directly into the groundwater model. The groundwater model can then calculate the groundwater chemical concentration as a function of unsaturated zone loadings

(simulated by PRZM), pesticide properties, and hydrogeologic conditions. The outputs of the groundwater model are the mean estimates of the concentration of chemicals in groundwater resulting from the use of different chemicals and tillage practices. However, estimates of mean concentration may not be useful for a risk analysis. Carsel et al. (1988) use a Monte-Carlo simulation technique to assess potential groundwater loadings of a chemical. To use this technique, probability density functions are determined, a Monte-Carlo simulation is run, and the predicted amounts of chemical residue are used to construct a cumulative probability distribution which, in turn, can be used to predict the percentage of simulations that resulted in residue movement below a certain soil depth.

Groundwater Loading Effects of Agricultural Management Systems (GLEAMS) Model Application

The following information describing the GLEAMS model application is derived from the work of Leonard and Knisel (1988), "Evaluating the Groundwater Contamination Potential from Herbicide Use." This example application demonstrates some of the functional relationships that can be developed with a chemical fate and transport model. Because GLEAMS models chemical movement only through the root zone, it also illustrates the application of a model that does not simulate chemical movement down to the saturated zone (unless the saturated zone is sufficiently shallow). The goal of this application is not to estimate chemical concentrations in groundwater, but rather to estimate relationships between chemical and environmental characteristics and leaching loss

(where leaching loss is defined as chemical that moves through the soil). The application consists of a comparison of leaching loss on two fields with different soil types in the Georgia Coastal Plains. A 50-year continuous simulation is used to estimate herbicide leaching losses when different chemical partition coefficients and half-life values are used. A chemical's partition coefficient and half-life are functions of the chemical's properties and the environmental characteristics. In this application, Leonard and Knisel do not present the amount of chemical leaving the root zone or the depths to which leaching occurs, although GLEAMS can estimate this. Instead, the authors determine the distribution of simulated chemical leaching losses. Leaching losses are expressed in percent of the application applied (50-year means are simulated).

According to Leonard and Knisel (1988),

> A difficulty encountered in applying continuous simulation models is the generation of so much specific detail that generalizations are difficult. A possible solution to this problem is to fit the specific model output to general relationships. . . . (p. 213)

Therefore, to compare the results from the two fields, the authors estimate the relationship between the leaching losses (as a percent of application) and a half-life/adsorption ratio. Both a log-linear and a linear functional form are used. Again, 50-year means are used; however, Leonard and Knisel also acknowledge that mean values may not be of interest in risk analysis. As an alternative to using a 50-year mean value, the authors use the annual pesticide leachate losses simulated for the 50-year period to express the percent of years that the amount of pesticide leached exceeds specified percentages of application. This

probabilistic approach may be more suitable when examining threshold-exhibiting health hazards.

Hydrological Simulation Program-
FORTRAN (HSPF) Model Application

The following information describing the HSPF model application is derived from the work of Bicknell et al. (1985), "Modeling Water Quality and the Effects of Agricultural Best Management Practices in the Iowa River Basin."

This HSPF application simulates water quality in the Iowa River Basin under a base scenario and a Best Management Practices (BMP) scenario. The BMP scenario consists of: (1) a shift from moldboard plowing to chisel plowing and field cultivation as primary tillage, (2) one summer cultivation for weed control instead of two, and (3) allowing crop residue to remain on the field following harvest. This application evaluates the water quality of the receiving body, the Iowa River at Marengo, Iowa. Because groundwater flows into this river, the contaminant contribution of shallow groundwater is simulated.

Alachlor, an herbicide, was used in this simulation; alachlor was chosen because it is used extensively on corn and soybean cropland in the basin. This simulation considered the following processes: application of the chemical to corn and soybean cropland, adsorption to soil, degradation, dissolved and adsorbed transport to streams, and instream adsorption, decay, and transport. The nutrient processes are also simulated including application of fertilizer and plant residue in chemical form to the land surface or incorporated into the soil, the soil nitrogen cycle, adsorption and desorption, nutrient movement with soil and water

to the stream, ammonia volatilization (loss of chemical into the atmosphere), and nitrification. The model parameters adjusted to reflect the effects of the BMP were soil moisture retention values, rainfall interception, surface roughness, land cover, and the sediment fines produced by tillage.

The simulation estimated edge-of-stream loadings for the base scenario and the BMP scenario. Edge-of-stream loadings of runoff, sediment, alachlor, nitrate, and ammonia were simulated. The total edge-of-stream loadings of runoff (mm), nitrate (kg/ha), and ammonia (kg/ha), is equal to the sum of surface runoff, interflow runoff, and groundwater runoff.

The results of this simulation are used to assess the risk or exposure of aquatic organisms to various magnitudes and durations of chemical concentrations. The simulations of alachlor concentrations under the base scenario and the BMP scenario are examined to determine the percent of time acute, chronic, and sublethal concentrations exist.

Conclusions

This chapter discussed some of the basic features of chemical fate and transport models and also presented some applications of these models. Management models were emphasized because they are most useful for policy analysis. For example, if a policy restricts herbicide use, producers may substitute a different type of tillage practice for the restricted herbicide. A model that considers these changes in management practices (such as the three models presented above) is necessary to estimate the change in potential groundwater pollution resulting from the implementation of an agricultural policy.

One major shortcoming of all three models is that only in certain situations can they be used to simulate chemical movement down to the groundwater. In situations where the groundwater zone is sufficiently deep, a gap exists between the lowest zone simulated (e.g., the bottom of the root zone) and the groundwater zone. Another shortcoming of these models is that they must be used to compare the relative effects of different management practices or scenarios; these models do not give absolute predictions of chemical loading (Leonard and Knisel, 1988).

CHAPTER 4

LINKING THE ECONOMIC AND ENVIRONMENTAL MODELS

In the previous chapters, economic production and chemical fate and transport were viewed as separate processes. This chapter uses stylized versions of physical and economic models to illustrate the linkage between these processes. Key to the linkage is the dynamic nature of both processes.

Consider first the dynamics of the environment. Damage-producing agents that are introduced into the environment eventually affect environmental quality. If the time elapsing between the introduction of these damage-producing agents and the manifestation of the damage is at least one production season, this relationship exhibits inter-seasonal dynamics and can be expressed as

$$a_t = f(a_{t-1}, z_{t-1})$$

where a is a vector of environmental characteristics and z is a measure of damage occurring (e.g., the amount of contamination leaching into groundwater). This assumption of inter-seasonal dynamics is a reasonable characterization of environmental dynamics because pollution is a cumulative process and usually becomes apparent in later seasons.

The vector of environmental characteristics, a, can be divided into two subvectors: (1) a_1, a vector of characteristics that affect

agricultural productivity and chemical transport, and (2) a_2, a vector of characteristics that do not affect productivity but do affect chemical transport.

The agricultural production process may exhibit intra-seasonal dynamics, inter-seasonal dynamics, or both, because a producer's optimizing decisions may include decisions made at the intensive margin and at the extensive margin. Decisions made at the extensive margin (e.g., what fields to put into production) are often inter-seasonal and therefore environmental characteristics may be considerations in these decisions. When inter-seasonal decisions are part of the agricultural production process, the producer's objective function can usually be specified as a multiperiod model. Decisions made at the intensive margin are often intra-seasonal (e.g., pesticide application rates) and the producer's objective function can usually be specified as a multistage model. Because changes in environmental quality resulting from agricultural chemical use may take a relatively long period of time to manifest themselves, environmental quality can be assumed to be constant over a production season. Therefore, for intra-seasonal decisions, environmental quality would be taken as given for that season.

To illustrate intra-seasonal decisions in a model, let m_t be a vector of management decisions made during season t:

$$m_t = g(m_{1,t}, m_{2,t}, \ldots, m_{n,t}).$$

The management practices of a field interact with the environmental characteristics of that field to create environmental damage; this physical relationship can be expressed as a damage function. The damage function may be a relatively simple relationship. For example, Nielsen

and Lee (1987) matched areas of "high" agricultural chemical use with areas of "high" DRASTIC scores. The damage function may also be as complex as a chemical fate and transport model.

The linkage between the agricultural production model and the environmental model requires some assumptions about the properties of the damage function (z_t) in relation to the vectors of environmental characteristics ($a_{1,t}$ and $a_{2,t}$). Depending on the environmental characteristics being considered, z_t may be a function of $a_{1,t}$, or $a_{2,t}$, or both. For example, if z_t represents potential chemical loading to groundwater, m_t represents chemical applications, $a_{1,t}$ represents soil quality, and $a_{2,t}$ represents groundwater quality, then z_t may be expressed as:

$$z_t = z(m_t, a_{1,t})$$

This relationship assumes that current groundwater quality does not affect the amount of current groundwater loading. However, there may be cases where z_t is a function of $a_{2,t}$. For example, if $a_{2,t}$ had a threshold level beyond which potential loadings are affected, then z_t would be a function of $a_{2,t}$.

To illustrate the linkage of economic and environmental models, consider the following set of relationships:

$$z_t = z(m_t, a_{1,t}) \qquad [4.1]$$

$$a_{1,t} = a(m_{t-1}, a_{1,t-1}) \qquad [4.2]$$

$$a_{2,t} = \bar{a}(z_{t-1}, a_{2,t-1}) \qquad [4.3]$$

Note that z_t is a function of m_t and $a_{1,t}$, and that $a_{2,t}$ is a function of z_{t-1}. These relationships are illustrated in Figure 4. Although the damage function does not affect the productivity-related environmental characteristics, management practices do.

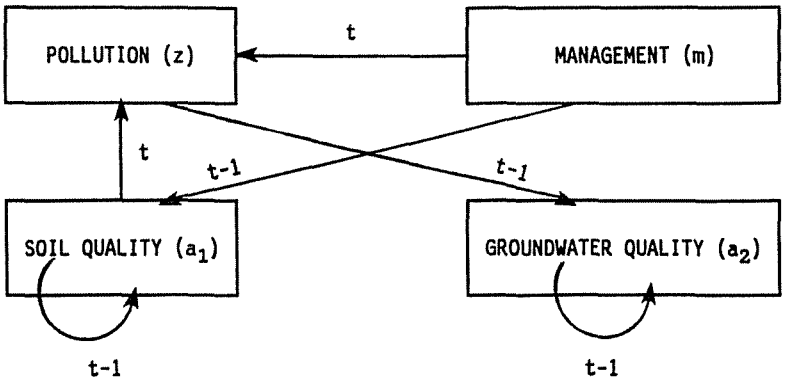


Figure 4. Relationships between economic and environmental models in the evaluation of groundwater pollution.

The economic and environmental models may be linked through the production function. As Figure 4 illustrates, only those environmental characteristics that affect productivity (the vector a_1) will be factors in the producer's optimizing decisions:

$$q_t = q(m_t, a_{1,t}). \qquad [4.4]$$

Substituting equation [4.2] into equation [4.4] yields

$$q_t = q[m_t, a_{1,t}(m_{t-1}, a_{1,t-1})] \qquad [4.5]$$

$$q_t = \bar{q}(m_t, m_{t-1}, \ldots, m_{t-n}). \qquad [4.6]$$

When $a_{1,t}$ is a function of past management decisions, output is a function of past and present management decisions. Thus, in the nomenclature of Chapter 2, the production process exhibits input dynamics.

If the vector $a_{1,t}$ is an argument of the damage function, the relationship between economic and environmental models can be expressed as a joint production function. Referring back to equation [4.1], the damage function can be expressed as:

$$z_t = z(m_t, a_{1,t}) \quad \text{and}$$

$$dz = (\partial z/\partial a_1)da_1 + (\partial z/\partial m)dm.$$

Assuming m_t represents chemical use and $a_{1,t}$ represents soil quality, one would expect the following relationships:

$$\partial z_t / \partial m_t > 0 \quad \text{and}$$

$$\partial z_t / \partial a_{1,t} < 0.$$

Assuming diminishing marginal product implies:

$$\partial^2 z / \partial m^2 < 0 \quad \text{and}$$

$$\partial^2 z / \partial a_1^2 < 0.$$

If the above relationships hold, the damage function can be presented as a series of positively sloped, concave isoquants as in Figure 5. This figure illustrates that while holding management practices constant, an increase in soil quality (an increase in a_1) will result in a decrease in environmental damage.

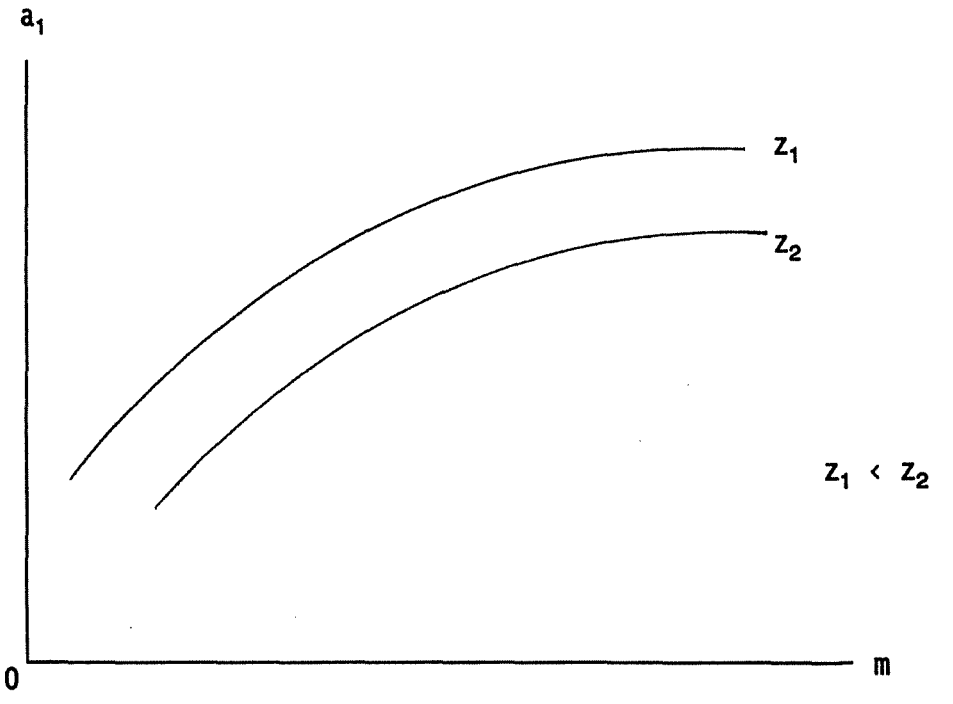


Figure 5. The damage model as a function of environmental characteristics and management practices.

To demonstrate that the production process jointly produces pollution and agricultural output, equation [4.1] can be inverted:

$$a_{1,t} = z^{-1}(z_t, m_t). \qquad [4.7]$$

Substituting equation [4.7] into equation [4.4] yields:

$$q_t = \hat{q}[m_t, z^{-1}(z_t, m_t)]$$

$$q_t = \underline{q}(m_t, z_t) \qquad [4.8]$$

Equation [4.8] illustrates the joint production of agricultural output and environmental damage. This joint production function depicts the combinations of output of the agricultural good and the amount of chemical loading to groundwater possible by varying soil quality (Figure 6). In Figure 6, m is being held constant and a is varied. As a changes, the resulting levels of q and z also change. This curve should be interpreted as an output-damage transformation surface and not as a production possibilities frontier because in this case variable inputs are held constant.

The position of this transformation surface within the quadrant is determined by m_t, the amount of chemical used during period t, and the producer's position along the transformation surface is determined by the soil quality. The shape of the isoquant depends on properties of the production and environmental damage functions. Assume that the transformation surface is concave as in Figure 6. If two producers used identical types and amounts of chemicals (m_t), the producer with relatively poor soil quality (e.g., soil conducive to groundwater pollution) would be operating at point B, whereas the producer with relatively good soil quality would be operating at point A. The producer at point A

obtains more output and less environmental damage than the producer at point B.

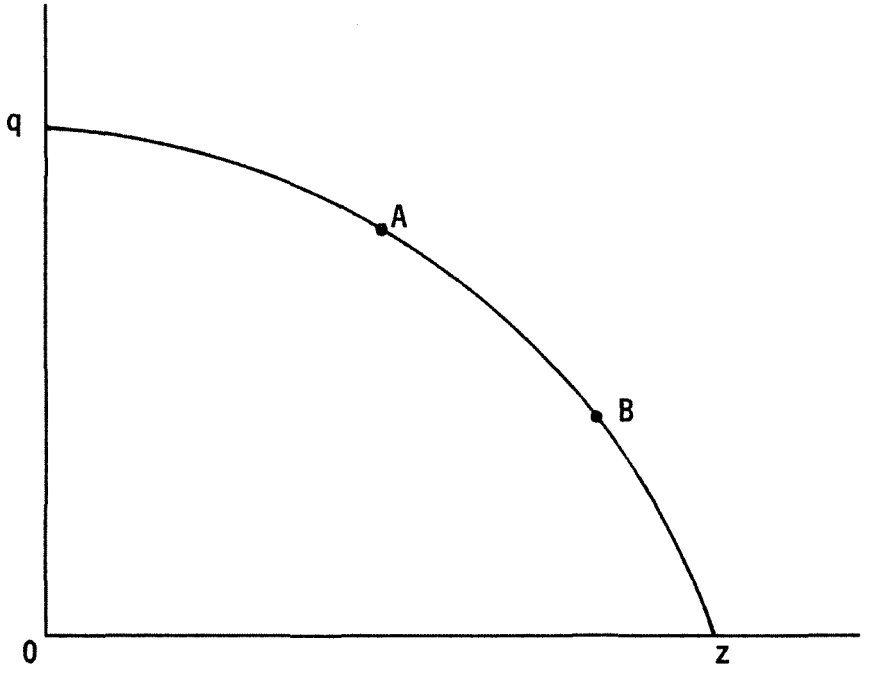


Figure 6. Output-damage transformation surface.

In this simplified model of economic and environmental interactions, the producer takes environmental damage into account only when a productivity-related environmental characteristic is being damaged. Assume:

$$a_{1,t} = \tilde{a}(z_{t-1}, a_{1,t-1}) \qquad [4.9]$$

$$q_t = q(m_t, a_{1,t}). \qquad [4.10]$$

Substituting equation [4.9] into equation [4.10] yields:

$$q_t = q[m_t, a(z_{t-1}, a_{1,t-1})]$$

$$q_t = \tilde{q}(m_t, z_{t-1}, a_{1,t-1}). \qquad [4.11]$$

In this case the costs of the changes in a_1 are being internalized by the producer, although any costs associated with changes in a_2 are not. However, as equations [4.1] through [4.6] indicate, if the environmental damage resulting from chemical use does not affect productivity, then the producer will not consider that environmental damage when making his production decisions. Both equations [4.6] and [4.11] indicate that all of the damage to the a_2 characteristics created by agricultural production is an externality.

One complication that can arise when linking the production and environmental models is the difference in time steps. The production model is constructed to express chemical use in terms of quantity of chemical applied at a point in time, whereas the chemical transport models are usually constructed to simulate changes over time. In order to reconcile the two models, the user may need to calculate the total chemical loading during a time interval meaningful to management decisions. For example, physical models can simulate change in chemical concentration per day or on a continuous basis (e.g., $\partial z/\partial t$), whereas the economic model examines the total change in chemical concentration over many days (e.g., z_t). The physical model results can be used in an economic model by calculating z_t as follows:

$$\int_{t_0}^{t_1} dz/dt \; dt = z(t_1) - z(t_0) = z_t$$

where t_0 is time at the beginning of the chemical model simulation, t_1 is the point in time being evaluated, and z_t is the total potential loading at time t.

So far, the level of aggregation of this simplified model has not been addressed. Although a deterministic model may be useful for a relatively homogeneous area of land, for other areas (e.g., larger areas), variations in environmental characteristics and production practices may be significant and a statistically valid description of the region is needed. Following Antle and Just (1989), a producer makes intensive and extensive production decisions based on input and output prices, policy factors, and individual farm characteristics (i.e., technology and risk attitudes of the producer). The extensive decisions, such as what fields to put into production, determine the environmental characteristics of land in production. The producer also makes intensive management decisions such as chemical use, tillage practices, etc. Therefore, the producers in an area are jointly determining management practices and environmental characteristics. The interactions of the management practices and the environmental characteristics then jointly determine the production of the agricultural output and of pollution for that area of land in production. Assume that

$$m = m(p, w, r) \qquad [4.12]$$

$$a_1 = a_1(p, w, r) \qquad [4.13]$$

where p is a vector of input and output prices, w is a vector of policy factors, and r is a vector of individual farm characteristics (i.e., technology and risk attitudes of the producer). Within a geographical region (and assuming perfect competition in factor and product markets), all producers will have identical price and policy vectors; only the individual farm characteristics will vary. Through producers' management and land use decisions, management practices and environmental

characteristics form a joint probability distribution. For this example, the joint probability distribution of management practices and environmental characteristics can be expressed as

$$\phi(m, a: p, w, r). \qquad [4.14]$$

Both output and environmental damage are functions of this joint distribution; the expected amount of damage can be expressed as

$$E(z) = \int z(m, a)\ \phi(m, a: p, w, r)\ dm\ da. \qquad [4.15]$$

Antle and Just (1989) present a log-linear production function and damage function that illustrate the effects of policy on the distribution of input use and resulting environmental damage. The damage function is expressed as

$$z = m^{\alpha}a^{\beta}, \qquad \alpha, \beta > 0 \qquad [4.16]$$

and the agricultural production function is expressed as

$$q = m^{\eta}a^{\nu}, \qquad 0 < \eta < 1. \qquad [4.17]$$

Assume that ln m and ln a are normally distributed and that z is log-normally distributed. Antle and Just (1989) demonstrate that the shape of the joint production function of q and z depends on properties of the production and environmental damage functions. Given the model presented in equations [4.16] and [4.17], the joint production function can be expressed as

$$\ln y = (\eta - \alpha\nu/\beta)\ln m + (\nu/\beta)\ln z$$

where a is varied and m is held constant. The shape of this joint production function depends on the value of ν.

Figure 7 presents possible shapes of this particular joint production function where a is varied and m is held constant. When ν is positive, an increase in a results in an increase in both q and z for any

level of m. If $1 > \beta-\nu > 0$, as a increases, z increases at a decreasing rate; if $\nu > \beta$ 0, as a increases, z increases at an increasing rate; if $\beta = \nu$, as a increases, q and z increase at an equal rate. When ν is negative, an increase in a will result in an increase in q and a decrease in z. This example illustrates that the amount of environmental damage occurring as a result of a policy will be determined in part by the relationship between output and environmental damage.

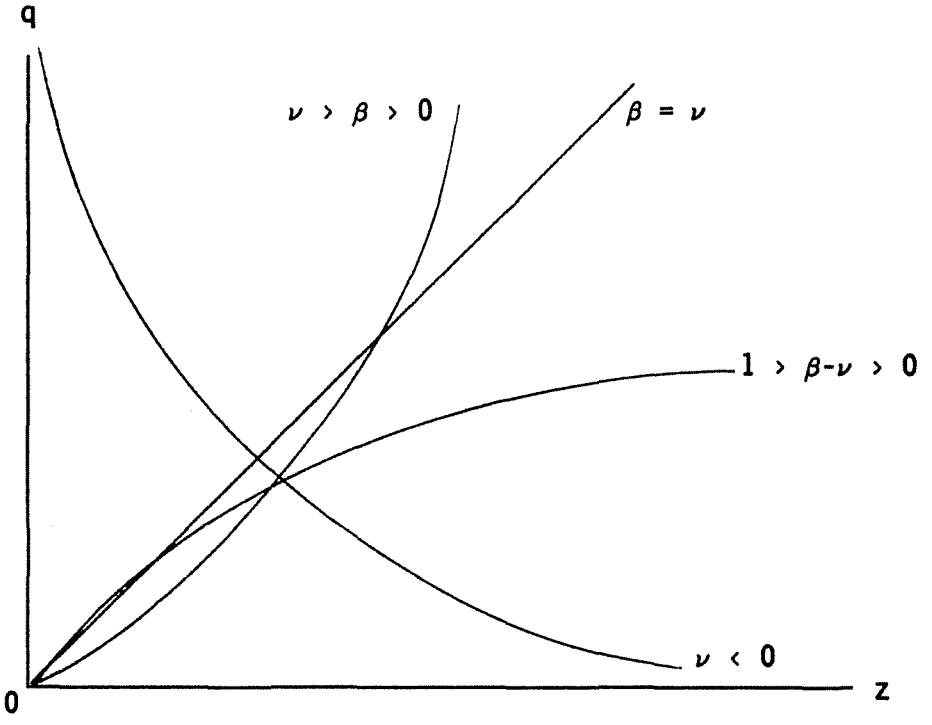


Figure 7. Output and damage relationships in the integrated log-linear physical-economic production model (Antle and Just, 1989).

A policy can influence a producer's production decisions at the intensive margin, the extensive margin, or both, which in turn affects the joint distribution of management practices and environmental

characteristics expressed in equation [4.14]. The amount of expected damage that results from a policy will depend on the changes in this joint distribution as well as the relationship between the productivity- and pollution-generating characteristics of a piece of land. For example, if highly productive land is more conducive to pollution than less productive land, a policy that encourages bringing highly productive land into production would result in an increase in expected damage.

To apply this statistical model to the analysis of potential loading of agricultural chemicals to groundwater, the management vector could represent chemical use (e.g., types of chemicals and amounts applied) and the vector of environmental characteristics could represent those characteristics important in determining an area's vulnerability to groundwater contamination (e.g., DRASTIC scores). (A DRASTIC score is a composite estimate of an area's potential vulnerability to groundwater pollution. The DRASTIC data base is discussed in more detail in Chapter 7.) Nielsen and Lee (1987) use a relatively simple application of this type of stochastic framework in a study that examines potential for groundwater pollution across the United States. The authors estimate the distribution of management practices (expressed in terms of "high" and "medium" levels of chemical application) and the distribution of environmental vulnerability to groundwater contamination (expressed in terms of "high" and "medium" DRASTIC scores). The joint distribution of management practices and environmental quality is estimated and the implied damage model is a simple relationship: an area with a high DRASTIC score and high chemical use would have a greater potential for

groundwater pollution than an area with a medium DRASTIC score and medium chemical use. This is a simple example of linking production practices (chemical use) and hydrogeologic characteristics (DRASTIC scores) to estimate groundwater pollution potential. Because Nielsen and Lee do not model the agricultural production process, the effects of the hydrogeologic characteristics and the damage function on producers' intensive and extensive production decisions are not examined.

CHAPTER 5

VALUING HEALTH RISKS

An important component in the determination of the social costs of groundwater pollution is the valuation of human health risks. This step can be relatively complicated because it involves the valuation of a non-market good, and because of the psychological considerations that accompany this type of valuation. A variety of methods have been used to determine the value of a human life. Mishan (1970) describes four commonly used methods of estimating the economic value of a life. The "gross output" approach estimates the loss to the economy as a result of an individual's death. This estimation consists of the discounted present value of the individual's expected future earnings. The "net approach" is similar to the gross approach; however, this estimation calculates the present discounted value of the losses over time accruing to others as a result of the death of the person at age "t". The third method uses the value that policy makers implicitly assign a life in the creation of policy. The fourth method, the insurance principle, assumes than an individual determines the value of his life based on the insurance premium he pays.

Mishan (1970) raises theoretical objections to all of the above methods, his strongest objection being that all four methods ignore the concept of potential Pareto improvement. Potential Pareto improvement

occurs when the sum of all individuals' compensating variations are greater than zero. Compensating variation is a measure of the change in an individual's welfare; it is the amount of money that can be taken away from an individual after an economic change, while leaving him as well off as he was before the change. For a welfare gain, it is the amount he would be willing to pay for the change. For a welfare loss, it is the amount he would be willing to accept as compensation for the change. Equivalent variation is another measure of welfare change; it is the amount of money we would need to give an individual if an economic change did not happen, to make him as well off as if it did. For a welfare gain, it is the compensation he would be willing to accept to forego the change. For a welfare loss, it is the amount he would be willing to pay to avoid the change.

Potential Pareto improvement occurs when the sum of all individuals' measures of compensating variation is greater than zero. The problem with measuring an individual's change in welfare is that when a person's life is threatened, his willingness to pay to avoid death (a measure of equivalent variation) may be bound by his income. Alternatively, an individual may decide that the amount of compensation necessary to persuade him to accept death (a measure of compensating variation) may be unbounded. This problem of potentially unbounded compensating variation and income-bound equivalent variation can be avoided by using the concept of the "statistical life" that considers only small changes in health risk. A project that changes the number of statistical lives saved does not save specific individuals, but rather changes the health risks or probability of death faced by a group of individuals. Using the

concept of a statistical life, the compensating variation can be used to measure the willingness to pay for a reduction in probability of death.

Mishan (1970) addresses another aspect that affects the valuation of life, the distinction between voluntary and involuntary risk. When evaluating the benefits of a voluntarily assumed service, risks associated with that service do not have to be valued because the individual using the service is already taking those risks into account. However, involuntarily assumed risk must be included in a benefit-cost analysis. To clarify this distinction, consider the case of a pesticide application. According to Mishan, any health risks the pesticides have on the pesticide applicator do not have to be valued because the applicator has already accounted for those risks when he evaluated the costs and benefits of using the pesticides. (This assumes that the applicator is aware of the risks.) The applicator's willingness to pay to use pesticides (his compensating variation) is net of the value of the health risks; to include the value of the health risks in a benefit-cost analysis would be to double-count the risks. However, the health risks faced by other individuals (e.g., the health risks attributed to pesticide-contaminated groundwater) should be included in the benefit-cost analysis because those individuals involuntarily exposed to the risks have not accounted for these costs in their private decisions.

Mishan (1970) advocates that the valuation of life should be based on a method that is consistent with the Pareto base of existing allocation theory and benefit-cost analysis, such as summing individual compensating variation. However, applying such a measure becomes difficult because of the psychological considerations that can influence

an individual's compensating variation. Weinstein and Quinn (1983) examine the normative value of health risk reductions, and present some of the psychological motives for society not implementing least-cost policies of lifesaving. In keeping with Mishan's (1970) results, Weinstein and Quinn illustrate that involuntarily-assumed risks may be psychologically less acceptable than voluntarily-assumed risks. Weinstein and Quinn also point out that the concept of blame and ethical responsibility may influence policy making; *ex ante* identification of victims may cause society to feel more responsible, and *ex post* identification may cause decision-makers to feel more accountable.

In the above discussion, health risks are perceived as being two separate commodities, voluntarily-assumed health risks and involuntarily-assumed health risks. However, it is possible that the framing or wording of the description of a commodity can result in an individual assigning different values to one commodity. For example, if an individual is asked to value a change in the level of provision of a commodity, and that commodity is framed two different ways (e.g., in terms of an increase in lives saved versus a decrease in lives lost), the individual may have two different values for that commodity. Framing effects can result from descriptions of the change in the level of provision of the commodity, as well as the method of payment for that change.

In normative decision theory, subjective probabilities of occurrence are supposed to be based only on beliefs about the likelihood of events; however, psychological research has found that these subjective probabilities reflect beliefs about both the likelihood and the individual's

preferences regarding the occurrence (a "wishful thinking" effect whereby a positive relation exists between the value of a consequence and its perceived likelihood of occurrence). Weinstein and Quinn (1983) also explain that, according to normative decision theory, choices and actions should follow from beliefs and values, although behavioral research indicates that the reverse may sometimes be true. This "pseudo-certainty" effect may also play a role in valuing health risk reductions; a project that gives the individual a feeling of certainty is given a greater value than the project that does not, even if the total (statistical) lives saved is the same for both projects.

Another important psychological determinant of an individual's valuation of a health risk reduction is the base level of risk involved. According to Weinstein and Quinn (1983) and Weinstein et al. (1980), the higher the base level of mortality probability, the higher the willingness to pay for a decrease in the mortality probability. In a later study, Smith and Desvouges (1987) find the estimated valuation of a risk change decreases with increases in the level of risk. This result would appear to be consistent with the "pseudo-certainty" effect.

The Smith and Desvouges (1987) study resulted in several important findings, including: (1) the estimated valuation of a change in the level of risk decreases with increases in the baseline level of risk, and (2) the valuation of a change in risk depends on the direction of the risk change. In the 1987 study, the individuals' mean bids to decrease risk were "uniformly greater" than the bids to avoid risk increases.

As an alternative to measuring compensating variation, Thaler and Rosen (1976) use a hedonic method to determine the value of saving a

statistical life. They establish a relationship between job risk and wage rates in order to impute a set of implicit marginal prices for various levels of risk. Thaler and Rosen then estimate the market wage-risk-personal characteristics equalizing difference function that matches workers and firms. The regression of this equation indicates a positively sloped wage-risk function, with the risk regression coefficient estimating the marginal wage rate paid to workers to assume another increment of risk. For example, if 1000 workers are employed at a job entailing an extra death risk of .001 per year, the yearly wage differential aggregated over all workers can be calculated, with this aggregate differential being the aggregate willingness to pay to save a statistical life.

Household production theory can be used to illustrate how chemical-related health risks affect producer decisions. Health can enter the economic production framework in several ways. Health effects may enter the production function directly if these effects decrease the farmer's labor productivity. For example, agricultural chemicals may have an effect on the farmer's health that diminishes his ability to work; this in turn affects the farmer's labor, a factor in the production function. Household production theory assumes households to be both producers and consumers. Becker (1971) shows that the cost per unit of a household-produced commodity, h_i, is the sum of the costs of purchased goods per unit and the "shadow" or opportunity cost of time:

$$\pi_i = a_i p_i + b_i w$$

where π_i is the total costs per unit of h_i produced, a_i and b_i are fixed input-output coefficients, p_i is the cost of purchased goods required to

produce a unit of h_i, and w is the amount of time necessary to produce one unit of h_i.

A change in an environmental variable (i.e., age, health, education) would change the amounts of goods and time required to produce a unit of h_i. The change in the environmental variable would change the efficiency of producing a unit of h_i by changing the input-output coefficients, a_i and b_i, thereby changing the marginal cost of h_i. For example, a farmer's health can enter the household production function as an environmental variable and can be incorporated as some probability of, for example, cancer or death. Agricultural production decisions can affect the producer's health, in turn affecting available labor. Because health is a factor in the household production function, health will also be a factor in the farmer's utility function.

This overview of methods and considerations of valuing human health risks illustrates that valuation is specific to the situation being considered. Smith and Desvouges (1987) showed that the level of baseline risk influenced valuation, as did the direction of change in the probability of health risks. Weinstein and Quinn (1983) explained some of the psychological factors affecting valuation. Their results indicate that extrapolation of survey results from one situation to another may not be appropriate. Therefore, to place a value on the health effects of groundwater pollution (using survey data), one would need to obtain data specific to that case of groundwater pollution.

Thaler and Rosen's (1976) method of determining the implicit value of saving a life could be modified to determine the value of the health risks resulting from agricultural chemicals. For example, one approach

would be to compare the wage differential of a pesticide applicator and a tractor operator, the differential being the value of the health risks of applying pesticides. A less precise approach would be to determine the upper bound on the value of the health risks. For example, if a chemical applicator could avoid the risk of health effects by using a gas mask, and if the applicator chooses not to wear one, then the price of the mask is the upper bound on the value of the risk; if the applicator does not use a mask, the value of the health risk must be less than the value of the mask.

The above methods deal with the health risks associated with the application of chemicals. The valuation of health risks associated with polluted groundwater may be more difficult because using wage differentials is no longer appropriate. One possible method of valuing groundwater health risks would be to use the value of a groundwater purification system as an upper bound on the value of health risks; this method was used by Nielsen and Lee (1987) in their valuation of the social costs of groundwater pollution. Alternatively, a hedonic approach may be used to determine a relationship between health effects and real estate values, the differential being the implied value of the risks. A problem with this approach is that one would expect land values to be measurably affected only when groundwater pollution levels are extremely high and land market participants know about the contamination and its effects. In the case of agricultural chemical pollution, these high levels may rarely occur. This technique may be useful in highly polluted situations, but referring back to the Smith and Desvouges (1987) study,

the values calculated could not be extrapolated for areas with different base levels of pollution.

As mentioned previously, a valuation method consistent with the concept of potential Pareto improvement involves determining individuals' compensating variations. These compensating variations (also called measures of willingness to pay, or WTP), can be elicited from individuals using survey techniques such as a contingent valuation (CV) survey. A CV is based on the concept that the elicited WTP bids are contingent on the situation presented to the survey respondents. An example of a CV survey will be presented in the case study in Chapter 8.

CHAPTER 6

HEALTH RISK ASSESSMENT

Merely determining the concentration of a groundwater contaminant may not be meaningful in itself. To make a level of pollution meaningful to people, the pollution can be expressed in terms of the health risks it presents. A health risk assessment can be used to place a value on a level of pollution or can be used for policy formulation; the Environmental Protection Agency uses risk assessment for chemical regulation. In order to quantify the human health risks presented by groundwater contamination, the types of health effects and extent of exposure must be identified. The potential health effects from exposure to agricultural chemicals include a variety of acute and chronic effects. Much of the exposure assessment research has concentrated on the hazards related to pesticide exposure, possibly because the hazards associated with a pesticide application are relatively easy to observe and to link to the chemical. For example, it may be easier to trace a case of chemical poisoning to a pesticide application than to groundwater contaminated by nitrogen fertilizer.

Moses (1987) outlines potential pesticide-related health problems faced by farmworkers, including cancers, reproductive abnormalities, and mental and psychological changes. Moses notes that the difficulty involved in determining the health risks of a pesticide is heightened

because of the "inert ingredients" in pesticides. Pesticides may contain both active ingredients and inert ingredients (ingredients that are not active against the target species); both types of ingredients may be toxic to humans, but the inert ingredients are not required to be tested for acute and chronic health effects or to be listed by name on the pesticide label. This makes quantifying the health effects of agricultural chemicals difficult because a complete analysis would consider all of the associated health effects, not just those caused by active ingredients.

In order to value the effects of agricultural chemicals on human health, the risks the chemical presents must be assessed. The National Academy of Sciences defines risk assessment as "the scientific activity of evaluating the toxic properties of a chemical and the conditions of human exposure to it in order both to ascertain the likelihood that exposed humans will be adversely affected, and to characterize the nature of the effects they may experience" (U.S. EPA, 1985, p. II-2). The Environmental Protection Agency (U.S. EPA, 1985) describes the four components used in a risk assessment:

(1) Hazard Identification: This component consists of gathering and evaluating data on the types of health injury or disease that may be produced by a chemical and on the conditions of exposure under which injury or disease is produced. It is also important to determine whether it is scientifically feasible to extrapolate toxic effects from one setting to another setting (e.g., extrapolating effects that occur in laboratory animals to humans). Information on the toxic properties of a substance can be obtained from animal

toxicology studies, controlled epidemiologic investigations of exposed human populations, chemical studies or case reports of exposed humans, experimental studies on isolated animal organs, cells, etc., and the analysis of the molecular structure of the substance. Results from animal toxicology studies and epidemiology studies are the two principal sources of toxicity data. The use of animal toxicity data assumes the effects on humans can be inferred from the effects on animals; this EPA publication asserts that this assumption has been shown to be generally correct. Also, Gough (1989a) demonstrates that estimates of human risk of cancer (associated with pollution) based on epidemiologic data and toxicologic data are similar.

(2) Dose-Response Evaluation: This evaluation determines the quantitative relationship between the amount of exposure and the extent of toxic injury or disease. In most cases, dose-response relationships are estimated from animal studies. The noncarcinogenic effects of a substance are generally believed to exhibit threshold effects; that is, there is a dose below which adverse effects will not occur. The dose-response evaluation for these noncarcinogenic effects involves describing observed dose-response relationships and determining experimental threshold levels, also called "no observed effect levels" (NOELs). The NOEL can be divided by some specified (but arbitrary) safety factor in order to determine the Acceptable Daily Intake (ADI) of the substance for humans.

Estimating dose-response relationships for substances that do not need to achieve threshold levels to cause adverse effects usually

requires the use of a mathematical model that estimates low-dose risks from high-dose risks. (It is generally thought that carcinogenic substances do not have thresholds, although there is some controversy regarding this point.) Estimating low-dose risks directly can be costly; research experiments can last years and a large number of observations is necessary. Regression analysis can be used to estimate a dose-response relationship by assuming the relationship has a particular functional form. Regulatory agencies currently use one-hit, multistage, and probit models, with regulatory decisions usually based on the one-hit or multistage model results. Multihit, Weibull, and logit models may also be used for risk assessment. Mehlman et al. (1979), Lave (1982), and Tardiff and Rodricks (1987) all provide descriptions of the models used to describe the dose-response relationship. Brown (1987) notes that although these models yield similar results over the range of observable response rates (relatively high-dose levels), extrapolating results from high-dose levels to low-dose levels is dependent on the mathematical form of the model used; the further the extrapolation from the observable response range, the greater the divergence of model results. Brown (1987) feels that this divergence of results is the greatest limitation of this extrapolation methodology and refers to mathematical dose response models as a "necessary evil" in the process of estimating low-level exposure effects.

None of the above mentioned dose-response models is chemical-specific; each is based on general theories of carcinogenesis rather

than on data for a specific chemical. It appears that modeling improvements will be brought about mainly through the incorporation of toxicological and biological information. "The contribution from statisticians and model-builders has reached an impasse, and more accurate extrapolations are not possible without additional information on the mechanisms of action of the toxic agents" (Brown, 1987, p. 265).

(3) Human Exposure Evaluation: This component describes the nature and size of the population exposed, and the magnitude and duration of the exposure. The exposure examined can be past, current, or anticipated exposure. To estimate the human dose of a contaminant, information regarding the dosage from various contact sites must be available. In the case of a water contaminant, potential contact routes include direct ingestion through drinking, inhalation, skin absorption from water (e.g., from bathing) and from contaminated soil, and from ingestion of contaminated soil. In order to evaluate dosage from a water contaminant, standard simplifying assumptions are often made (e.g., an adult is assumed to ingest two liters of water per day). The total dose an individual receives is equal to the sum of the doses from all possible routes. However, the concept of absorption can complicate this estimation; the dose that enters the body is not always equal to the dose that is absorbed. If the absorption rate of a substance is not known, the rate is often assumed to be 100 percent.

(4) Risk Characterization: Using information from the above components, risk characterization determines the likelihood that humans will

experience any of the various forms of toxicity associated with the substance. For noncarcinogenic effects, the margin of exposure (MOE) is used as a surrogate for risk; it is the experimental NOEL divided by the estimated daily human dose. The larger the MOE, the smaller the risk. For carcinogenic effects, risk per unit of dose is estimated from the dose-response modeling. (If a variety of assumptions and models is used, a range of risks might be produced.) To estimate total risk, the actual human dose is multiplied by the risk per unit of dose.

A health risk assessment requires an understanding of the hazards a particular substance presents. As mentioned above, epidemiologic studies can be used to estimate potential health risks. An epidemiologic study examines the incidence, distribution, and control of disease in a population. Many of the epidemiologic studies on the toxicity of agricultural chemicals focus on the most easily observed health effects and the most direct route of human exposure. For this reason, there is a relatively large number of epidemiologic studies on acute pesticide poisoning, often due to direct contact with the pesticide (e.g., exposure while hand-applying the chemical).

Loevinsohn (1987) presents an epidemiologic study that examines the relationship between insecticide use and increased mortality. His study area is Central Luzon in the Philippines, where organophosphate and organochlorine insecticides have been widely used by farmers. Loevinsohn compared mortality rates for the rural population (the exposed group) and the urban population (the control group) in two periods, one of low use of insecticides (1961-71) and one of high use (1972-84). In Central

Luzon, pesticides are usually applied by men using backpack applicators; protective clothing is generally not worn.

The study results indicate a possible linkage between mortality and occupational exposure to insecticides, with mortality increasing only in the age and sex class occupationally exposed (men aged 15-54 years). The overall increase in non-traumatic mortality among rural men between periods of low and high insecticide use was 27.4 percent, from 2.15 to 2.74 per 1000 (a 0.59 per 1000 increase). Also, causes of mortality likely to be confused with insecticide poisoning (i.e., stroke) also increased in rural men (aged 15-54) between low-use and high-use periods.

If it could be established that insecticide use does increase the applicator's risk of mortality by 0.00059, a hedonic approach similar to that used by Thaler and Rosen (1976) could be used to determine the aggregate willingness to pay to save a statistical life. Unfortunately, it is often difficult to establish causality from epidemiologic studies. One reason (as this study illustrates) is the difficulty in implementing controls in epidemiologic studies. Loevinsohn (1987) is able to establish that there may be some association between high levels of insecticide application and increased mortality, but no conclusive associations can be made because of the lack of an adequate control group.

On a similar note, Berteau and Spath (1986) point out, "A study conducted by the California Department of Health Services (Jackson et al., 1982) is the only known actual epidemiological study that has been conducted on people exposed to water contamination with pesticides" (p. 429). The study indicates a possible link between a soil fumigant

(DBCP) in drinking water and stomach cancer, but once again the lack of an adequate control group makes the determination of conclusive associations impossible.

As indicated by the above studies, health risk assessment can be complicated by a number of factors. A tradeoff exists in the determination of human health hazards. Human health hazards can be extrapolated from laboratory animal studies; in this case, control groups can be established but extrapolating human hazards from animal hazards may not be appropriate. Estimating human health hazards from epidemiologic data eliminates the extrapolation problem, but introduces the possibility of inadequate control groups. Also, synergistic effects between substances may be important in estimating health hazards. These are just a few of the factors that may add to the imprecision of a risk assessment of contaminated ground water.

There is still much to be discovered about the human health effects of groundwater contaminated by agricultural chemicals. However, as Gough (1989b) points out, agricultural chemical contamination of groundwater is a relatively insignificant cause of human cancers. He states, "Only a small percentage of human cancers is caused by the environment in the sense of air, water, soil and their contaminants" (personal communication). Current sunlight exposures, environmental tobacco smoke, and indoor radon account for between 70 and 96 percent (depending on which approach is used to calculate total cancer risk) of all cancers associated with environmental exposures. This indicates that groundwater pollution by agricultural chemicals does not present a major cancer risk to the public in general. It should be noted that some groups of people

are more susceptible to the health risks presented by agricultural chemicals; for these people, the health effects of contaminated groundwater may be significant.

CHAPTER 7

DATA BASES

In order to perform any type of empirical analysis of groundwater contamination, some combination of data on environmental quality (a), management practices (m), and groundwater contamination (z) is needed. The suitability of a data base can be evaluated only after the type of analysis to be performed is determined. In the evaluation of potential groundwater pollution, data can be used for three general levels of analysis:

(1) Descriptive Analysis: This type of analysis can qualitatively evaluate pollution potential based on environmental characteristics and management practices. The data used may be relatively aggregated. For example, Nielsen and Lee (1987) use this descriptive approach; in their study they employ DRASTIC scores (scores that are an aggregate measure of hydrogeologic quality), and combine them with agricultural chemical usage to qualitatively determine groundwater pollution potential. The end result of their study is the classification of counties according to the level of chemical application ("high" or "medium") and the level of hydrogeologic quality ("high" or "medium").

(2) *Ex Post* Analysis: An *ex post* analysis requires data on the incidence and levels of groundwater contamination, environmental

characteristics, and management practices. For example, a study could be undertaken to determine the relationship between the amount of groundwater contamination that has occurred and the properties of the chemicals applied (e.g., solubilities and degradation rates).

(3) *Ex Ante* Analysis: Several of the chemical fate and transport models were designed to be used for *ex ante* analysis. For example, PRZM was designed to be used in pesticide registration. Of the three types of analysis, *ex ante* usually has the largest data requirement. As discussed in Chapter 3, the user is usually required to input data on hydrogeologic characteristics, soil characteristics, management practices, and chemical properties. Additionally, an *ex ante* analysis may require calibration of some parameter values, increasing the data requirement. Although a simulation of changes in groundwater quality will require a relatively large amount of data, it may be less costly to conduct (in terms of time, money, and health risks) than a survey of groundwater contamination (as would be required for an *ex post* analysis).

This section surveys some of the data bases currently available, with a focus on national data bases.

Environmental Quality Data Bases

DRASTIC Data Base

DRASTIC is a system developed by the National Water Well Association (NWWA) to evaluate the potential for groundwater pollution of hydrogeologic sites in the United States. This system was designed to fulfill several requirements. The system must: (1) function as a management

tool, (2) be simple and easy to use, (3) utilize available information, and (4) be able to be used by individuals with diverse backgrounds and levels of expertise (Aller et al., 1986). As with the chemical transport models discussed, the DRASTIC system is intended to be used on a comparative basis; the DRASTIC county scores can be used to estimate a county's vulnerability relative to the vulnerability of another county.

The DRASTIC system consists of two main sections: (1) the designation of hydrogeologic settings, and (2) the application of a system for relative ranking of hydrogeologic parameters to evaluate the relative potential for groundwater pollution of the hydrogeologic setting. A hydrogeologic setting is a composite description of all the major geologic and hydrologic factors which affect and control groundwater movement into, through, and out of an area. It is defined as a mappable unit with common hydrogeologic characteristics and therefore similar potential for groundwater pollution (Aller et al., 1986). DRASTIC uses a classification system that divides the United States into 13 groundwater regions: (1) Western Mountain Ranges, (2) Alluvial Basins, (3) Columbia Lava Plateau, (4) Colorado Plateau and Wyoming Basin, (5) High Plains, (6) Nonglaciated Central Region, (7) Glaciated Central Region, (8) Piedmont and Blue Ridge, (9) Northeast and Superior Uplands, (10) Atlantic and Gulf Coastal Plain, (11) Southeast Coastal Plain, (12) Hawaiian Islands, and (13) Alaska. These groundwater regions are then classified into smaller hydrogeologic settings. Approximately 86 hydrogeologic settings have been identified within the groundwater regions of the United States.

The second portion of this evaluation system involves the relative ranking of hydrogeologic parameters called DRASTIC factors. These factors are considered to be the most important mappable factors determining groundwater pollution potential. These seven factors form the acronym DRASTIC: D, depth to water; R, (net) recharge; A, aquifer media; S, soil media; T, topography (slope); I, impact of the vadose zone; C, (hydraulic) conductivity of the aquifer.

To estimate the relative groundwater pollution potential of a hydrogeologic setting, these seven factors are rated. Because these factors have unequal impacts on groundwater vulnerability to pollution, the factors are weighted. There are two indices of weights: the generic DRASTIC index and the Agricultural DRASTIC index. The Agricultural DRASTIC index was developed to examine the application of agricultural pesticides, whereas the generic DRASTIC index is used for all other contaminants, including fertilizers. The agricultural index assigns higher weights to topography and soil media. The weighted DRASTIC score for a hydrogeologic setting is calculated by multiplying each factor rating by its respective weighting and then summing these products.

The Environmental Protection Agency (EPA) incorporated the DRASTIC rating system into its National Pesticide Survey (NPS). NPS will be described in more detail later in this section. Stage I of the NPS consists of the classification of all U.S. counties in terms of groundwater vulnerability. Research Triangle Institute (RTI), under contract to EPA, completed this first stage. A modification of the DRASTIC system was used to classify all 3,144 counties of the United States into three

categories of groundwater vulnerability. Alexander et al. (1986) describe the procedures and results of this study.

After obtaining the DRASTIC factor data, the numerical DRASTIC scores were computed, the most important scores being the weighted score and the VARSCORE. The VARSCORE represents the weight score and an index of variability; because the weighted score does not account for heterogeneities in the characteristics of the seven DRASTIC factors, an "index of variability" can be used to reflect these heterogeneities. In terms of the Agricultural DRASTIC score, two factors, aquifer media and the impact of the vadose zone, are thought to be the primary factors where variability may apply (Alexander et al., 1986).

In this RTI application of the DRASTIC system, the classifications of vulnerability are based on a 10/60/30 percent distribution of the data base of VARSCORES for low, medium, and high vulnerabilities, respectively. This distribution was selected to minimize the likelihood of counties being misclassified as having either high or low vulnerability.

The authors of this application note that the DRASTIC system has several shortcomings. Because the area of study (the county) is relatively large, the averaging effect may result in areas of high vulnerability within a county being overshadowed by areas of low vulnerability within the same county. Also, rating DRASTIC factors is generally subjective and differences in opinion may occur when rating these hydrogeologic factors over county-sized areas.

As mentioned earlier, Nielsen and Lee (1987) use weighted DRASTIC scores calculated by RTI to conduct a descriptive analysis of groundwater pollution potential from agricultural chemicals. In that study, the two

main data bases used to determine distributions of groundwater vulnerability and agricultural chemical usage are the DRASTIC index and the Resources For the Future (RFF) National Pesticide Use Inventory data base.

Interactive Soils Information System (ISIS)

ISIS consists of a group of programs designed by the U.S. Army Construction Engineering Research Laboratory to access and analyze USDA/ Soil Conservation Service soil data. The two data bases in this system are the Soil Interpretive Records (SOI-5) data base and the Map Unit Records (SOI-6) data base.

SOI-5 contains data on over 16,000 different soil series nationwide. A soil series is a group of soil horizons that are similar in differentiating characteristics such as texture, color, structure, consistence, reaction, content of carbonates or other salts, content of organic matter, and minerological composition. Each SOI-5 record contains information including estimates of soil properties such as soil texture, erosion, flooding, pH, organic matter, permeability, depth of bedrock, frequency and duration of flooding, yield estimates of crops, suitability or limitations of soils for specified land uses, and soil features affecting specified land uses. Data for SOI-5 is obtained from SCS Soil Interpretation Records that are completed by soil scientists. The data base is accessed by three programs: (1) the Soils Information Retrieval System (SIRS), (2) the Line Printer Soils Information Retrieval System (LPSIRS), and (3) the Multiple Parameter Series Search (MPSS) (Thompson et al., 1987).

SOI-6 contains data on more than 175,000 soil mapping units nationwide. [According to Thompson et al. (1987), a taxonomic unit is a named kind of soil (taxon) that has specific properties with defined limits or ranges in characteristics; a soil map unit is an area of soil(s) delineated on a soil map that contains one or more taxonomic units. A soil mapping unit is the aggregate of the map units of a kind of soil(s) identified by the same name (symbol) on a soil map or in a soil survey area.] The data on the soil mapping units includes mapping unit characteristics, critical phase criteria, and survey acreage by county. These data are obtained from SCS Map Unit Records. The programs that access the SOI-6 data base are the Map Unit Use File System (MUUFS) and the Computer-Aided Land Evaluation System (CALES).

One shortcoming of ISIS is that neither of the two data bases, SOI-5 and SOI-6, provide information concerning where within a geographical area a map unit or soil series occurs. However, SCS state and local offices can provide exact locational information (Thompson et al., 1987).

Management Data Bases

A variety of management practices can affect the amount of groundwater contamination occurring. Two of the practices that can have the greatest impacts on contamination potential are chemical use and tillage practices; therefore, this section focuses on chemical use and tillage practice data bases.

Fertilizer Use Data Base

The Economic Research Service (ERS) Division of the USDA has developed a fertilizer use data base from survey data for the primary producing states of corn, cotton, soybeans, and wheat. The data is for the state level and the fertilizers considered are nitrogen (N), phosphate (P_2O_5), and potash (K_2O). The data information includes percent of acres fertilized of the acres surveyed in each state, the average application rate for those acres receiving fertilizer, and the percent of proportion fertilized at or before seeding, after seeding, and at both times. Detailed data on fertilizer use by other crops are not collected in any consistent manner.

Pesticide Use Data Base

The Economic Research Service (ERS) of the USDA has developed a pesticide use data base from survey responses. The data are for the regional level. (USDA has divided the U.S. into 10 production regions.) The crops included in the data base are: wheat, corn, sorghum, soybeans, cotton, peanuts, rice, tobacco, barley, oats, pasture and rangeland, alfalfa, and "other hay," and the data are expressed in pounds of active ingredients applied. This survey has been conducted several times in the past two decades, but in 1989 the survey format was changed. The new survey consists of state-level data for wheat, cotton, soybeans, and corn. Instead of data on application rates, data on numbers and percentages of acres treated are included. According to Herman Delvo of the ERS (personal communication, February 1989), application rates do not

need to be included because these rates do not vary significantly over time.

National Pesticide Use Inventory

This data base, developed by Resources For the Future (RFF), is a more complete pesticide data base than that of the ERS. The RFF data base includes estimates of the use of 15 soluble pesticides by state for 10 major field crops. These 15 pesticides represent approximately 53 percent of the total national usage of the 55 active ingredients monitored in EPA's National Pesticide Survey. The 15 pesticides represented are: alachlor, metolachlor, atrazine, cyanazine, 2,4-D, trifluralin, carbaryl, disulfoton, methyl parathion, carbofuran, diazinon, dinoseb, acifluorfen, chlorothalonil, and fluometuron (Gianessi, 1987). The 10 major field crops (representing approximately 90 percent of the national volume of use of the 15 pesticides) are: corn, soybeans, wheat, barley, oats, sorghum, cotton, rice, sugarcane, and hay. Estimates of use of each pesticide are made on a state-by-state and crop-by-crop basis, with an assessment being made for each state that has a significant acreage of any of the 10 field crops (Gianessi, 1988). Although county-level estimates of pesticide use are contained in the data base, these estimates are extrapolated from state-level estimates; the assumption is that there is uniform use within a state for the same crop/pesticide combination.

As discussed in an earlier section, the amount of active ingredient applied to a field may not be a good measure of the amount of potentially harmful chemicals applied to that field. "Inert" ingredients may present

more potential health risks than active ingredients. Therefore, when using a data base that measures chemical applications in terms of amount of active ingredient applied, it should be remembered that the amount of potentially hazardous chemical applied may be underestimated.

Tillage Data

The Conservation Technology Information Center (CTIC) has developed the CTIC National Survey of Conservation Tillage Practices. Data have been collected each year from 1983 through the present. The survey elicits information on the use of five conservation tillage practices (no-till, ridge-till, strip-till, mulch-till, and reduced-till) for 12 major crops for every county in all 50 states. Crops included are: corn (full season and double crop), small grains (spring seeded and fall seeded), soybeans (full season and double crop), cotton, grain sorghum (full season and double crop), newly established forage crops, newly established and renovated permanent pasture, and "other crops" (a catch-all category for all other planted acres in a county). The information in this data base is aggregated to the county level.

Contamination Data Bases

Groundwater Contamination Data Base

Algozin et al. (1988) have developed the Groundwater Contamination (GWC) data base to compare the groundwater contamination potential of various agricultural practices under various environmental conditions. This national data base includes both environmental quality information and management practices information. The data base represents the

compilation of six files: the 1982 Natural Resource Inventory (NRI) file (land use information); the Soils-5 Interpretation file (soil and water table characteristics; three 1985 Resource Conservation Act (RCA) budget files (production practice and economic data); and the DRASTIC index file.

The data base can be separated into two macro files which are joined via several linkage variables. The macro file that describes the environmental characteristics of an area is the foundation of the data base and consists of the NRI file, the SOILS-5 file, and the DRASTIC file. The 1982 NRI consists of approximately 700,000 records of specific sample points where the data is collected; each record includes information on the physical characteristics of the site, erosion potential, and vulnerability to groundwater contamination. In this data base NRI data are used for some point-specific data, such as soil characteristics, and DRASTIC scores (county averages) are used for other characteristics such as rainfall. The SOILS-5 data base has been described earlier.

The second macro file provides the economic production information relating to each sample point using data from the three RCA budget files. This macro file is called the RCA Midset and was developed by the Center for Agricultural and Rural Development (CARD) at Iowa State University. The information contained in the RCA Midset includes: crop rotation, tillage, conservation practices, and input uses and yields for all crops included in the rotation.

Algozin et al. (1988) use this GWC data base to link up areas of groundwater vulnerability with areas where agricultural practices conducive to groundwater contamination occur. Because their analysis

incorporates site-specific information (from the NRI file) when computing DRASTIC scores, these scores are a significant improvement upon the county DRASTIC scores used by Nielsen and Lee (1987) that use only county-averaged data.

Pesticides in Groundwater Data Base

The Pesticides in Groundwater Data Base, developed by the EPA Office of Pesticide Programs, is a compilation of individual groundwater monitoring studies for pesticides. EPA intends to use the data base to identify pesticides that are contaminating groundwater, to identify areas that are vulnerable to groundwater pollution, to develop a better understanding of the chemical migration pathways and environmental processes associated with groundwater contamination, and to identify gaps in current data.

The data contained in this data base are from numerous studies conducted by federal and state agencies, chemical manufacturers, and universities. Approximately 130 reports have been entered into the data base. In addition to collecting and entering these studies, EPA has been conducting a program to assess the validity of the information gathered. To date, only 50 percent of the reports have been confirmed. Because the studies in the data base are from numerous sources, there is no consistency in methodology. Also, these studies have a variety of focuses; some studies concentrate on more vulnerable (i.e., more shallow) groundwater and other studies include wells that are not necessarily used for drinking water. (Consequently, this data base is not appropriate for estimating human exposure.) Two other shortcomings of the data base are

that many agricultural areas of the U.S. have not been sampled and therefore are not represented in the data base, and that establishing the source of pesticide contamination is difficult. According to Williams et al. (1988), determining the source was the most difficult parameter to confirm in the data base. Consequently, Williams et al. stress that the data base is to be used as a screening device and that "derived parameters contained in the report and in the appendices should be treated as indicators rather than absolute values" (p. 4-2).

The data base is separated into three data bases, with the Pesticide Monitoring Data Base containing the bulk of the pesticide information including: the pesticides tested for in the study, the suspected origin of the pesticide, number of wells tested/number of positive wells found, number of samples tested/number of positive samples found, lowest concentration found, highest concentration found, average concentration of positives, and a data confirmation rating (that indicates the quality of the data).

WATSTORE

WATSTORE, a Water Data Storage and Retrieval System, was established by the U.S. Geological Survey in 1971 to modernize the USGS water-data processing and management. Currently, files are maintained for the storage of: (1) surface-water, quality-of-water, and groundwater data measured on a daily or a continuous basis; (2) annual peak values for streamflow situations; (3) chemical analyses for surface and groundwater sites; (4) water-data parameters measured more frequently than daily; (5) geologic and inventory data for groundwater sites; and (6) summary

data on water use (Kilpatrick, 1981). Although WATSTORE contains six major files, the two files of primary interest for groundwater contamination are the Ground-Water Site-Inventory (GWSI) file and the water quality file. The GWSI file contains information on wells and springs at sites from all 50 states. As of February 1981, the file contained data for nearly 700,000 sites. The GWSI file is composed of hydrologic, geologic, and well inventory data for groundwater sites including site location and identification, geohydrologic characteristics, well construction history, and one-time field measurements such as water temperature. The data base is a combination of county-averaged data (e.g., climate), and point-specific data (e.g., soil properties); however, the data are statistically valid at the Major Land Resource Area (MLRA). There are approximately 169 MLRAs in the U.S. An MLRA is characterized by relatively homogeneous soils, crops, and production practices. Although the bulk of GWSI data is collected by USGS personnel in cooperation with federal, state, and local government agencies, the data may be biased in that those wells suspected of contamination are sampled.

The water-quality file contains data on the analyses of water composition. The file contains a relatively large amount of data on nitrate analyses of water, but relatively little data on pesticide analyses. This is because nitrate levels are somewhat easily and cheaply determined and so nitrate analyses are routinely performed. Pesticide analyses, however, are more difficult and expensive to conduct and consequently are often performed only as part of a special project. Although some areas of the country have assembled more pesticide data

than others, on the whole, nitrate information is more extensive than pesticide information. WATSTORE contains data on nitrate levels in samples collected from 87,000 wells throughout the U.S. during the past 25 years.

National Pesticide Survey

EPA has developed a National Pesticide Survey (NPS) to obtain a better understanding and characterization of agricultural chemicals in drinking water wells. This survey focuses on the quality of drinking water in wells and will elicit data on the relationship between contaminated well water and groundwater. The two major objectives of the NPS are:

(1) to characterize nationally the distribution of pesticide residues in community system and rural household water wells, and

(2) to assess the associations between the agricultural use of pesticides, the distribution of agricultural pesticide residues in well water, and hydrogeologic factors that influence groundwater contamination (Alexander et al., 1986).

The NPS is "the first nationwide survey of pesticide contamination in domestic and community water wells in the United States" (USEPA, 1987, p. 1). The survey involves sampling approximately 1,500 drinking water wells, and the survey results are expected to be statistically representative of over 13 million domestic wells and some 51,000 community water systems. Fifty-five active ingredients will be monitored in this survey. Because a goal of the survey is to determine the relationships among agricultural pesticide use, environmental characteristics, and well

water contamination, this survey involves more than simply testing well water. The survey includes interviews with householders and community water system operators, and data collection in the area surrounding each well. The information elicited will include the uses and characteristics of the well water, well construction data, and information on factors that could explain the source or route of contamination (e.g., chemical spills, abandoned wells).

Prior to the survey, EPA developed health advisories for the 61 pesticides with the highest leaching potential. These advisories were designed to be used by well owners, operators, and the general public to determine whether the contamination levels found in well water warrant further action. EPA also prepared non-technical summaries of the health advisories to explain the health risks presented by exposure to these pesticides. (These health advisories and summaries could be used in valuing the health risks of pesticides.)

CHAPTER 8

CASE STUDY

A case study is presented in this chapter in order to evaluate the feasibility of linking together the chemical fate and transport, economic, and human health models. Sugar beet production in the Yellowstone Valley of Montana was selected for this analysis because this production process includes irrigation and a relatively high use of agricultural chemicals, two factors that may contribute to groundwater pollution.

To begin this case study, knowledge of the production practices is necessary. Baquet and Johnson (1988) present a study on the costs and returns of irrigated crops in the Yellowstone Valley. The irrigated crops are sugar beets, malt barley, alfalfa, and corn silage; these four crops are usually used in some type of rotation. The counties included in this study are Treasure, Big Horn, Yellowstone, Carbon, Stillwater, and Rosebud. Baquet and Johnson (1988) conducted surveys of 15 randomly sampled producers in this six-county area who produce sugar beets, and determined production practices, costs, and returns. Survey results are grouped into three farm sizes: small (less than 100 acres of sugar beet allotment), medium (between 100 and 250 acres of sugar beet allotment), and large (more than 250 acres of sugar beet allotment). Baquet and Johnson formed a composite estimate of the production practices, costs, and returns for each of the three groups. The composite results of the

surveys of the medium-sized farms are presented in Table 1. The authors point out that the composite estimates "should not be interpreted as representative for any particular farm in the Yellowstone Valley, but rather a composite estimate of the producers surveyed" (Baquet and Johnson, 1988, p. 1).

An understanding of the general production practices involved in sugar beet production is necessary to get an idea of some of the types of decisions facing the producer; also, production practices (e.g., chemical use and types and timing of tillage) are inputs in management-oriented chemical fate and transport models.

Seedbed preparation for sugar beet production starts in the late summer or in the fall with the plowing of fields. Fields must be leveled before planting if surface irrigation is to be used. In the Yellowstone Valley, planting takes place between late April and late May. Baquet and Johnson's composite estimates indicate that fields are cultivated about three or four times per season, and are irrigated about four or five times per season. Plants must be thinned to 10 to 12 inches apart in a row. Although thinning can be done both mechanically and by hand, most of the producers surveyed used hand-thinning methods. Harvesting sugar beets consists of three general steps: (1) topping the plant, (2) lifting the root, and (3) loading them.

Because sugar beets are a relatively profitable cash crop, they are usually given the priority position in a crop rotation; that is, rotation away from sugar beets is done mainly to control sugar beet diseases and nematodes (Chapman and Carter, 1976). It is because of this necessary rotation that it would be difficult to isolate the potential chemical

Table 1. Production operations and materials for sugar beets.

OPERATION	EQUIPMENT		TRACTOR OR OTHER POWER UNIT	TIMES OVER	LABOR		MATERIALS USED	AMOUNT PER ACRE	UNIT	PRICE PER UNIT
	TYPE	SIZE			HIRED	FAMILY				
Plow		4-16"	140 hp	1						
Roller Harrow		15'	140 hp	2						
Plane		12'	105 hp	2						
Fertilize			Custom App.	1			140 # N 90 # P 30 # K			
Triple K		17½'	105 hp	1-2						
Plant		6 Row	75 hp	1			Seed	2 #/A.		$15/cwt.
Cultivate		6 Row	105 hp	4			Roneet	12 #	#/A	$1.10/#
Spray		6 Row	75 hp	1			Betamix Poast	1 qt/A. 1 qt/A.		$6.6/gal. $8/gal.
Thin				1	X					$40/A
Irrigate				4		X				
Defoliate		6 Row	105 hp	1		X				
Dig		3 Row	140 hp	1		X				
Truck 1					X					
Truck 2					X					
Truck 3						X				

(SOURCE: Baquet and Johnson, 1988, p. 6.)

loading attributable directly to sugar beet production. For a long-term simulation of potential groundwater loading, the effects of all of the crops in the rotation must be considered.

Sugar beet production is relatively chemical intensive, requiring both high levels of fertilizer and pesticide use. Moderate sugar beet yields such as those obtained in the Yellowstone Valley (about 20 tons per acre), can remove large amounts of nitrogen, phosphorus, and potassium from the soil. Fertilizer can be applied before seeding, during seeding, or as topdressing during the growing season. Timing of applications is important for both fertilization and irrigation. Fertilizer applications should be timed so that soil nitrogen is depleted 10 to 14 days prior to harvest (to encourage sugar production in the root). Table 1 indicates that fertilizer use per acre consists of: 150 pounds nitrogen, 90 pounds phosphorus, and 30 pounds potassium. Timing of irrigation is also crucial; the final irrigation of the season should meet the plant's water requirements, but should also ensure that the soil is dry enough for harvesting, yet not so dry that it impedes digging the beets.

The most common weeds in sugar beet fields are pigweeds, lambs-quarters, barnyard-grasses, mustards, kochias, and wild oats (Chapman and Carter, 1976). Weed control is accomplished with the use of herbicides, mechanical cultivation, and hoeing. Mechanical cultivation can be substituted for chemical weed control up to a point; however, once the sugar beet plants reach a certain size, mechanical cultivation between the rows is no longer possible. The three pesticides included in Table 1 (Ro-Neet, Betamix, and Poast) are all herbicides. It should be noted

that nematodes and rip-maggots can cause significant sugar beet damage in the Yellowstone Valley, although nematicides and insecticides are not mentioned in Table 1. Hallenbeck and Cummings-Burns (1985) provide a good description of the various classes of pesticides, their acute-exposure human health effects (observed in humans), their chronic human health effects, and their expected human health effects (which may be extrapolated from animal studies). Much of the following information on the chemicals Ro-Neet and Betamix is taken from Hallenbeck and Cummings-Burns (1985). Cycloate, a synonym for Ro-Neet, is in the thiocarbamate class of chemical compounds. Effects observed in humans following exposure to cycloate include: coughing, dermal irritation, eye irritation, and respiratory mucous membrane irritation. Suspected effects may include paralysis, dermal edema, and dermal hyperemia. Cycloate may also present significant chronic effects, although EPA has not yet made its studies available to the public. The active ingredients in Betamix are phenmedipham and desmedipham, which are in the carbamate class of pesticides. The potential chronic exposure effects of carbamates include anorexia, cholinesterase depression, muscle weakness, and renal damage.

The chemical fate and transport model used in this application is PRZM. Although a computer simulation was not run, a description of potential sources of key model parameters is included. It should be noted that the development of the input data set is no small feat. Carsel et al. (1984) divide a model simulation into five tasks and indicate the relative effort required for their completion. Of the five tasks, the development and input of the data set is estimated to require 40 percent of the effort. The type of data necessary for the analysis

will depend on the level of aggregation. Although several of the data bases discussed in Chapter 7 may provide the necessary data for the use of simple screening methods, such as the calculation of a DRASTIC score, acquiring the necessary data for a more site-specific analysis using a numerical model is more complex. The *PRZM User's Manual* (Carsel et al., 1984) does provide many estimates of model parameters as well as techniques to calculate parameters, but the level of disaggregation may not be sufficient for a site-specific analysis. What follows is a discussion of some of the key model parameters and some of the potential data sources.

The Montana Agricultural Potentials System (MAPS) is a data base of Montana hydrogeologic characteristics that contains information that can be used for this case study. This data base sections the entire state into eight-square-mile blocks; data on several dozen climatic and hydrogeologic characteristics are gathered for each block of land. The data include characteristics on the growing season, land use, potential evaporation, potential evapotranspiration, precipitation, regions, and soils. Precipitation characteristics include monthly averages and 24-hour precipitation. Soil characteristics include general soils of Montana (136 map units), soil water holding capacity, soil temperature, soil depth classes, soil pH, and a rainfall intensity factor.

Pesticide Parameters

As discussed in Chapter 3, the fate of a pesticide applied to soil depends largely on two of the pesticide's properties: persistence and

solubility. These properties are taken into account in PRZM by the pesticide transformation rate and partition coefficient.

(1) Pesticide Transformation Rate: Transformation rates are often obtained from chemical-specific studies. It should be noted that compared to surface transformation rates, relatively little is known about chemical transformation rates below the soil surface. It is not uncommon to estimate a subsurface rate as a percentage of the surface rate. Based on limited laboratory data, Donigian and Carsel (1987) estimated transformation rates in the horizon below the plant root zone and in groundwater to be 50 percent of the rate in the surface soil.

(2) Chemical Partition Coefficient: As discussed in Chapter 3, a chemical partition coefficient (K_p) is a major determinant of the leaching ability of a chemical. The K_p for each soil horizon can be expressed as a function of the chemical's adsorption coefficient for organic carbon, K_{oc}, and the organic carbon fraction or organic matter in each soil horizon. The Soil Conservation Service collects some data on organic carbon content of soil horizons, and the Soils Information Retrieval System (SIRS) contains data on soil organic matter. According to Leonard et al. (1987), data on the K_{oc} are not available for all pesticides, but significant work has been done to provide methods of estimation. These methods are based on a relationship between K_{oc} and K_{ow} (octanol: water distribution coefficient), and between K_{oc} and water solubility (see Kanaga and Goring, 1980, and Rao and Davidson, 1980).

Soil Parameters

Key soil parameters of PRZM include field capacity, wilting point, and the percent of sand or clay in the soil.

(1) Field Capacity and Wilting Point for All Horizons Modeled: Although the wilting point will depend on specific soil characteristics (e.g., organic matter content), most soils will have a similar wilting point for all common plants. Carsel et al. (1988a) obtained data on wilting point, field capacity, bulk density, and organic matter from the Soil Conservation Service. Bulk density is the mass per unit of undisturbed soil, dried to constant weight at 105°C; organic matter consists of the remains and decomposition products of both plants and animals (Fitzpatrick, 1983). MAPS contains soil field capacity data; also, Carsel et al. (1984) present three possible field capacity estimation techniques.

(2) Percent Sand or Clay: SIRS contains these data.

Hydrologic and Crop Characteristics

The hydrologic and crop characteristics determine the amount of water available for infiltration and runoff. Key parameters include precipitation, erosion, SCS curve number, evapotranspiration, depth to groundwater, and recharge.

(1) Precipitation: MAPS includes data on monthly averages, peak 24-hour precipitation rates, and a rainfall intensity factor, and also expresses monthly precipitation as a percentage of annual precipitation.

(2) Erosion: A modified USLE is used to estimate soil erosion. This equation requires information including peak runoff rate and the soil erodibility factor. MAPS contains both peak 24-hour precipitation rates and a rainfall intensity factor. The soil erodibility factor is a soil-specific parameter and specific values can be obtained from the SIRS data base or from the local Soil Conservation Service office.

(3) SCS Curve Number: The SCS curve number technique is used to estimate runoff. This technique accounts for the interaction of the hydrologic soil group, land use, and treatment (cover crop). Carsel et al. (1984) provide a method for calculating the SCS curve numbers and include some of the data needed for this calculation; however, the data included are limited to the major agricultural crops in the U.S.

(4) Evapotranspiration: MAPS provides data on both potential evaporation and potential evapotranspiration.

(5) Depth to Groundwater: Although the ISIS does contain some data on the depth to the water table, many PRZM applications rely on region-specific studies by the U.S. Geological Survey.

(6) Recharge: Recharge is the amount of water per unit of land that penetrates the ground surface and reaches the water table (Nielsen and Lee, 1987). Sources of net recharge data include the U.S. Geological Survey, State Geological Surveys, Soil Conservation Service, and State Department of Natural/Water Resources (Alexander et al., 1986).

One important note regarding data sources is that many of the chemical fate and transport model applications examined rely on studies of specific chemicals or specific sites for data, instead of on large, national data bases.

PRZM was designed to simulate pesticide loading to a small area where soil and climatic characteristics are relatively homogeneous. Because this case study examines the pesticide loading potential of this six-county sugar beet producing area of the Yellowstone Valley, a Monte-Carlo simulation technique can be used to account for the heterogeneity of soil and climatic characteristics in this area. In their study of aldicarb application to Ohio corn, Carsel et al. (1988a) incorporate several randomized parameters, including those for weather year, pesticide degradation rate, and values of field capacity, wilting point, and organic matter for each soil horizon. (The distributions of field capacity, wilting point, and organic matter were identified using SCS soil series data.) Once the input data set is complete, a series of Monte-Carlo simulations can be run and the predicted amounts of chemical movement in the simulations can be used to construct a cumulative probability distribution that indicates the number of simulations that resulted in leaching below the root zone. The results of this series of Monte-Carlo simulations can be used to compare simulation results under different management scenarios. As noted in Chapter 3, many management models are designed to be used to compare the potential loadings of alternative management practices. More importantly, for policy analysis we are interested in evaluating the effects of a change in policy because

a change in policy can affect production decisions, possibly changing the distribution of potential loadings.

As an example of a policy that affects management decisions at the intensive margin, assume that a policy is implemented that restricts the use of a herbicide, say Ro-Neet. This type of change could affect both chemical use and tillage practices. As discussed in Chapter 4, the types of weed control methods that will be substituted for Ro-Neet will be determined by the producer's economic production function. To determine the amount of potential loading resulting from the change in policy, the PRZM model parameters affected by the policy change must be identified and the parameters must be adjusted to reflect the changes in production practices. Assume that the restriction on the use of Ro-Neet results in producers using more mechanical cultivation. The change in chemical use can be incorporated into the PRZM input data set by adjusting the rate (and possibly timing) of chemical application. The change in tillage practice may involve a change in the amount of tillage or the use of a different type of tillage.

Once the input data set has been adjusted to reflect the changes in management practices, a second series of Monte-Carlo simulations can be run to determine the effect of the change in policy on the distribution of potential loading. The simulation results can be used to construct a cumulative probability distribution similar to that of the pre-policy series of simulations. From the cumulative probability distribution, it is possible to establish the percentage of simulations for which the model predicted residue movement less than (or greater than) a specified amount and below a specified depth. (For this case study the relevant

depth is the bottom of the root zone.) It is also possible to determine the percentage of simulations that predicted residue movement (exceeding a specified amount of chemical) below the root zone.

A shortcoming of PRZM is that it simulates pesticide movement only in the plant root zone. To determine the amount of pesticide that enters groundwater, it may be necessary to link up PRZM with a model that simulates pesticide movement and transformation down through the unsaturated zone. To determine the groundwater concentration of a chemical, a groundwater model must be linked to the saturated and unsaturated zone models. Linking these models can be a difficult procedure that is complicated by a lack of data on some of the physical, biological, and chemical processes occurring below the root zone. Carsel et al. (1988b) provide a relatively simple method of converting a potential pesticide loading to a groundwater concentration, although more theoretical groundwater models like the Analytical Transient One-, Two, and Three-Dimensional (AT123D) model (Yeh, 1981), are more complicated. If the unsaturated zone is sufficiently shallow, the potential loading simulated can be used directly in a groundwater model, and the potential groundwater concentration of the chemical can be calculated.

Valuation of the Social Costs

In this case study, a contingent valuation (CV) method is proposed to elicit information that can be used to value human health risks. A CV survey presents survey respondents with a hypothetical scenario and asks them to value a change in the level of the provision of some public good (or to avoid a change in the level of a public good). The

hypothetical scenario allows respondents to consider both use and nonuse (existence) benefits in their valuations and, because the scenario presented is hypothetical, a CV survey can be used for an *ex ante* valuation of a change in the provision of a public good.

A CV elicitation can obtain the appropriate compensated measure (holding the individual's level of utility constant) associated with a change in the level of the provision of the public good. This allows the individual to make his own tradeoffs between money and the public good. The bids elicited from a CV survey can be expressed as the difference between two expenditure functions. The compensating surplus (CS) can be expressed as:

$$CS = e(p_0, y_0, U_0) - e(p_0, y_1, U_0) \qquad [8.1]$$

where p_0 is the vector of prices, y_0 is the initial level of provision of a public good, U_0 is the individual's initial level of utility, and y_1 is the new level of provision of the public good (Mitchell and Carson, 1989).

The difference between the two expenditure functions (CS), is the amount of money the individual would be willing to pay to change from y_0 to y_1, while holding his utility level constant at the initial level, U_0. An advantage of the CV method over other methods of valuing public goods (e.g., a hedonic price method) is that the CV method does not require the estimation of an implicit demand curve. However, a disadvantage of the CV method is that it is subject to a variety of biases that may result because of: (1) incentives for individuals to misrepresent responses, (2) values being implied in the survey, and (3) scenario misspecification

by the researcher or misperception of the scenario by the respondent (Mitchell and Carson, 1989).

In this analysis of agricultural chemical policy, we want to evaluate the changes in the social costs and benefits resulting from a change in agricultural chemical use and possibly a change in tillage practices. It is strongly suggested that this change in the social costs and benefits (a change in the level of human health risk) can be valued with a contingent valuation method (Mitchell and Carson, 1989). However, in this case study, the chemical fate and transport model used (PRZM) simulates chemical movement only down through the root zone and not down to groundwater. Clearly it would not be meaningful to ask survey participants to value a change in the concentration of the chemical at the bottom of the root zone. Therefore, a gap exists between the lowest sector modeled by PRZM (the root zone) and the horizon that would be relevant to survey respondents (groundwater). This problem could be resolved by linking PRZM with a model that simulates chemical fate and transport from the bottom of the root zone to the groundwater zone. As mentioned in Chapter 3, work is being performed to develop a vadose zone model that would provide this linkage. For the purposes of this case study, we will assume that the unsaturated zone is sufficiently shallow that PRZM can simulate potential groundwater loadings. These loadings can be input into a groundwater model to estimate the chemical concentrations, and these concentrations can be used in a CV survey. Although the assumption that the depth to groundwater is sufficiently shallow may not be realistic for the Yellowstone Valley, it may be a realistic assumption for other areas in the U.S. (e.g., parts of Florida).

To use the CV method, a survey tailored to the situation being examined must be developed. As discussed in Chapter 5, each CV is particular to a specific situation, and so extrapolating CV results from one situation to another is not appropriate (see Smith and Desvouges, 1987, and Weinstein and Quinn, 1983). Furthermore, in order to elicit meaningful responses, the scenario presented to the survey respondents must be meaningful. According to Mitchell and Carson (1989),

> . . . the principal challenge facing the designer of a CV study is to make the scenario sufficiently understandable, plausible, and meaningful to respondents so that they can and will give valid and reliable values despite their lack of experience with one or more of the scenario's dimensions. (p. 120)

Nielsen and Lee (1987) state that the primary potential effects of agricultural chemicals in groundwater are human health risks. To illustrate the necessary components of the development and implementation of a CV survey, an example of a survey that elicits willingness to pay (WTP) values for reductions in groundwater contamination (of Ro-Neet) is presented. The survey will examine a reduction in the risk of dying of cancer (25 years in the future) as a result of exposure to this chemical via groundwater.

Chapter 5 addressed some of the psychological motives that can influence a contingent valuation of a change in a health risk. Conducting a CV on a reduction of the risks of dying of cancer presents some additional risks. Whereas many health risk valuations present the potential risks in terms of annual risks, the risks associated with cancer are not readily classified as annual risks. Cancer usually manifests itself many years after exposure to the cancer-causing agent

and so the risks of dying of cancer should be presented to the survey respondents as risks to be incurred in the future. Also, the level of exposure necessary to cause cancer can be very low and, according to Mitchell and Carson (1986), "Research has shown that people have difficulty understanding abstract risk levels and that they tend to exaggerate low-level risks" (p. 7).

The goal of this CV survey is to elicit WTP bids for a reduction in the level of groundwater contamination. The change in the level of groundwater contamination may have little meaning in itself; it is the associated change in the health risks that is meaningful to respondents. Therefore, the risks associated with the various levels of chemical contamination to be valued by the respondents must also be presented. Chapter 6 discussed types of data that can be used to determine the risks presented by various levels of chemical contamination. (Data on the toxicologic effects of a specific pesticide may be available from the Health Effects Division, Office of Pesticide Programs, Environmental Protection Agency.) It was also mentioned in Chapter 6 that there are several routes of exposure to water-borne chemicals. Data on the potential health effects of agricultural chemicals are incomplete, making it difficult to specify a dose-response relationship for low-level exposure to a chemical. However, as Nielsen and Lee (1987) point out, although the risks associated with low-level exposure may be uncertain, the public's perception is that they are significant. An individual's willingness to pay (WTP) is dependent on perception of risk, not necessarily actual risk.

Communicating the idea of low-level risks may require that the potential risks of the contaminated groundwater be compared with more familiar low-level risk (i.e., the risk of being struck by lightning). In their CV, Mitchell and Carson (1986) use two risk ladders to familiarize respondents with the low-level risks involved. The first risk ladder illustrates a full range of risks including basic risks (e.g., the risk of dying associated with different age groups), and specific "special" risks (e.g., risks associated with some professions). A second risk ladder that provides more detailed information on the lower-level risks is also presented. In addition to presenting a variety of relatively familiar low-level risks, this ladder also compares the number of cigarettes an individual would have to smoke in a lifetime to run the same risk of dying of heart disease or cancer as the individual would have of dying as a result of one of the low-level risks. Illustrations of these risk ladders are presented in Figures 8 and 9.

A referendum situation can be used to elicit survey participants' WTP for a reduction of the chemical concentration in groundwater. A referendum has the advantage that people may be familiar with the referendum method because decisions regarding the provision of public goods are often made this way. An open-ended referendum format can skirt the problem of starting point bias, but it can also produce a large number of nonresponses or protest bids. "Starting point bias occurs when the respondent's WTP amount is influenced by a value introduced by the scenario" (Mitchell and Carson, 1989, p. 240). However, an open-ended format can work well if the concept of the CV scenario is familiar to the respondents. For example, a CV survey could ask respondents how much

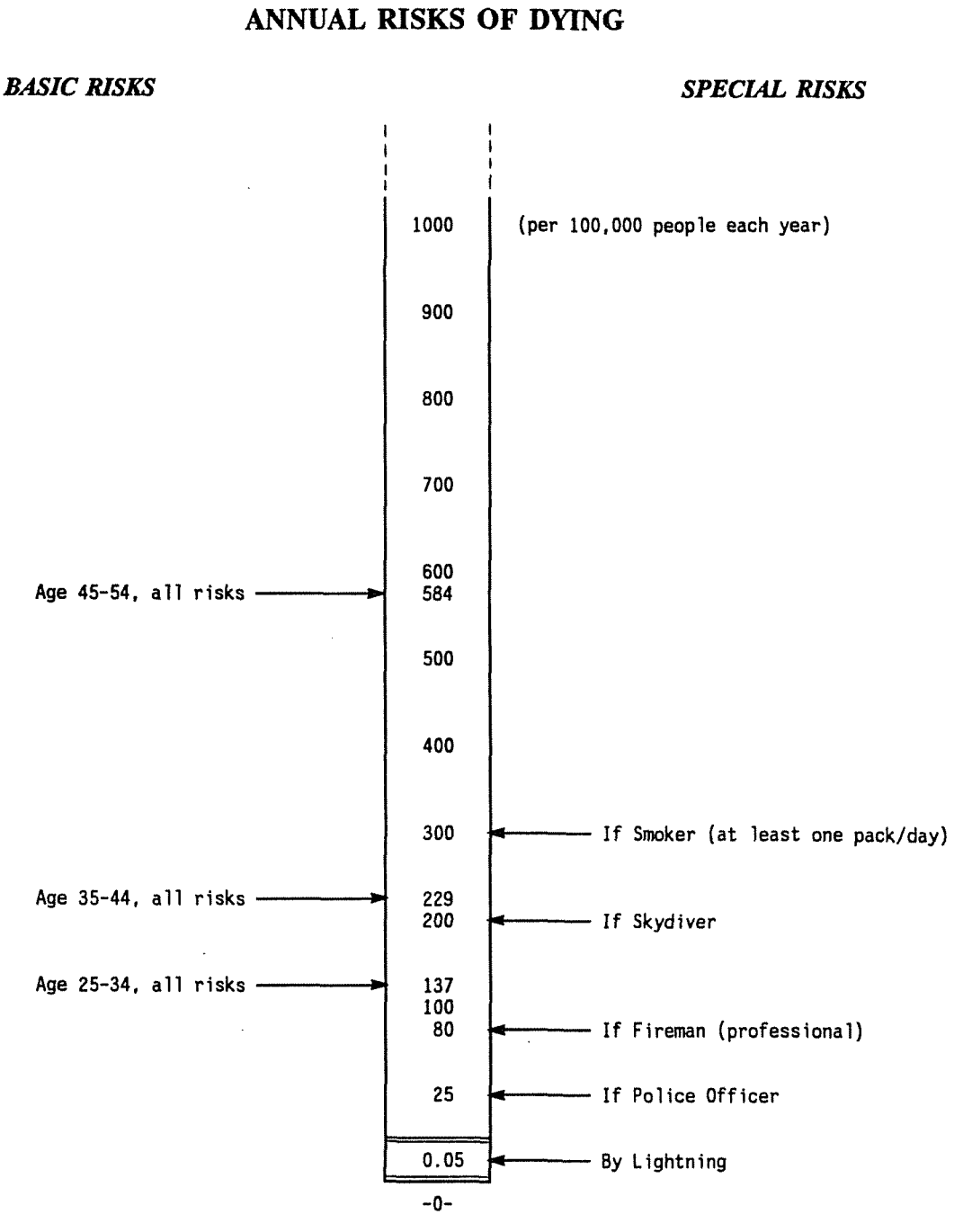


Figure 8. Risk ladder illustrating annual risks of dying (Mitchell and Carson, 1986, p. 46).

LOWER-LEVEL RISKS OF DYING
(annual)

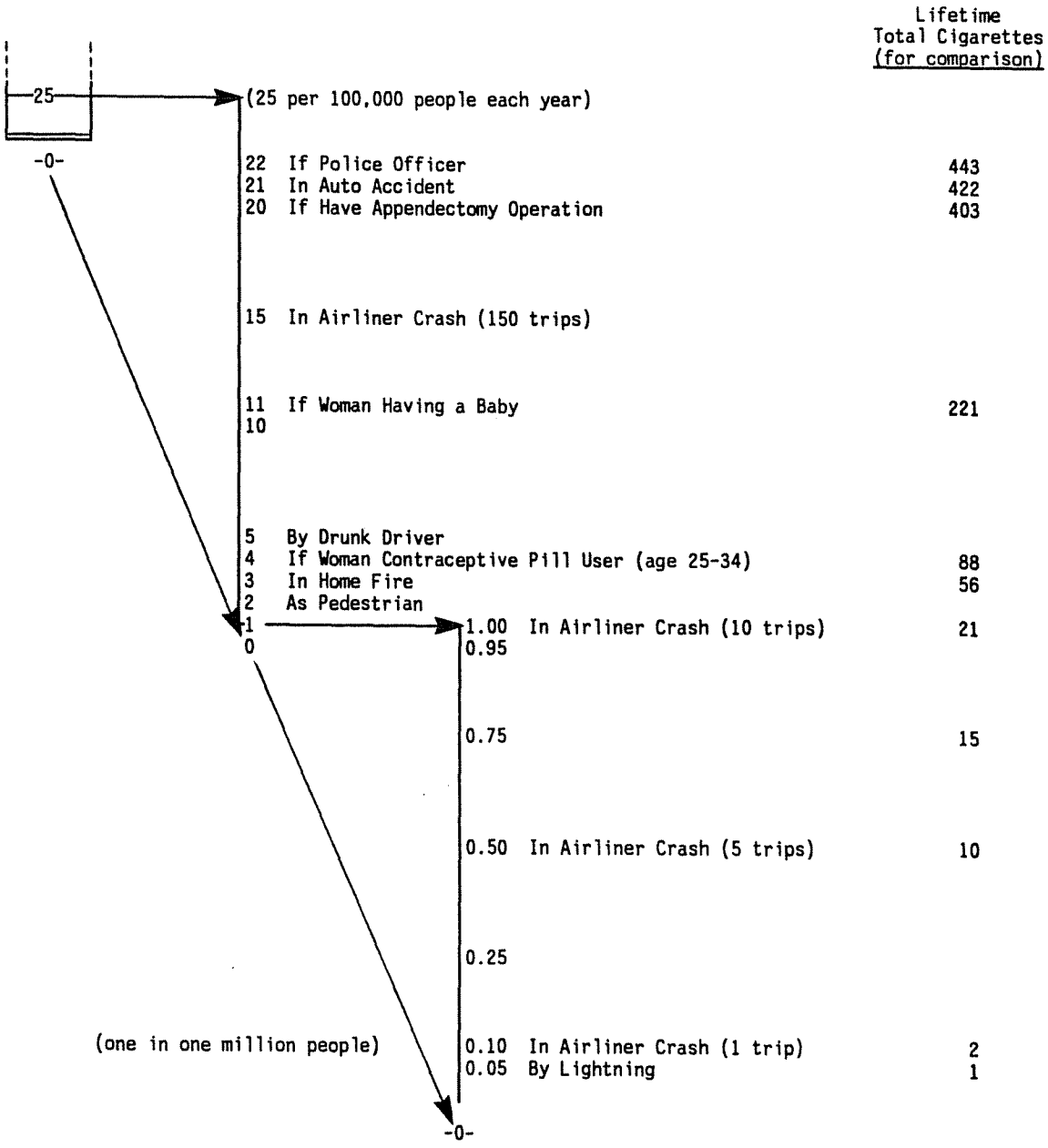


Figure 9. Risk ladder illustrating lower-level risks of dying (Mitchell and Carson, 1986, p. 47).

they would be willing to increase their water bills. If the respondents are familiar with the concept of paying water bills, then the number of nonresponses may be less than if respondents were not familiar with the concept of paying water bills.

For the purposes of this example CV survey, assume that the Yellowstone Valley population relies solely on a public water supply that originates as groundwater. The survey will be designed to elicit WTP bids for a reduction in the groundwater concentration of Ro-Neet as a result of purifying the water supply. Survey participants will be shown two risk ladders similar to those of Mitchell and Carson (1986), but since cancer risks are not readily classified as annual risks, this case study's risk ladders would present a variety of risks of dying prematurely (e.g., an individual's risk of dying prematurely if his occupation is that of a race car driver). As with Mitchell and Carson's risk ladders, the bottom portion of the first risk ladder will be enlarged and will be used in a second risk ladder to illustrate the low-level risks involved in this case study scenario. This second risk ladder will present a variety of low-level risks of dying prematurely (e.g., risk of being killed by lightning, risk of dying during an appendectomy operation). The levels of risk of dying prematurely of cancer presented by the two relevant groundwater concentrations of Ro-Neet (z_1 and z_2) will also be included in this risk ladder. Determining the human health risks associated with levels of groundwater concentration of a specific chemical may be difficult, as was the case with groundwater concentrations of Ro-Neet. Gosselin et al. (1984) indicate that for a "moderately toxic" pesticide such as Ro-Neet, the probable oral lethal dose for a 70

kg. person is a dose of 0.5 - 50 gm./kg.; this wide range of values of lethal dose may be meaningless on a risk ladder. This case study looks at the change in the risk of dying of cancer, but Gosselin et al. (1984) do not provide information on the dose-response relationship for chronic effects such as cancer. Although EPA may have this type of information, it is not currently available to the public.

Assuming that the dose-response relationship was available, the CV survey could be conducted as follows. One adult per household surveyed will be asked if they are willing to pay higher water bills (to cover the costs of water treatment) in order to reduce chemical concentrations in groundwater from z_1 to z_2 (where $z_1 > z_2$). Those who respond positively will then be asked for the highest amount they would be willing to pay for this reduction in groundwater concentration.

Once a CV survey has been conducted, the resulting data can be used to estimate the value of a statistical life (VSL). Mitchell and Carson (1986) present several methods of calculating the VSL; two of these methods are presented below:

(1) Calculate the implied VSL from the mean and median WTP values. WTP can be expressed as:

$$WTP = \delta \times VSL$$

where δ is the probability of a health risk occurring. Therefore, VSL can be calculated by dividing each of the WTP bids by its respective risk reduction. Because in this case study WTP bids of households (not individuals) were elicited, WTP bids would be divided by the respective risk reduction and by the size of the household surveyed.

(2) Use individual observations to estimate VSL. This method is based on the calculation of compensating surplus (CS) as expressed in equation [8.1]. To use this method, Mitchell and Carson express the initial expenditure function as:

$$E_0 = E[p_0, q_0 | U_0^*,\ R_0(AS,HB,OS,SEX),\ \delta=0,$$
$$T(AS,AT,ED),\ HS] = INC_c,$$

where p_0 is a vector of current prices; q_0 is a vector of currently provided public goods; $R_0(\bullet)$ describes the household's initial level of risk as a function of the household's age structure (AS), habits (HB), occupational structure (OS), and SEX; $T(\bullet)$ describes taste as a function of the household's AS, attitudes (AT), and education (ED); and HS is a measure of household size.

Changing the level of risk but holding all other arguments constant yields a new expenditure function, E_j:

$$E_j = E[p_0, q_0 | U_0^*,\ R_0(AS,HB,OS,SEX),\ \delta=\delta_j,$$
$$T(AS,AT,ED),\ HS] = INC_j.$$

The compensating surplus is the difference between these two expenditure functions and can be expressed as:

$$CS = f[INC_0,\ R_0(AS,HB,OS,SEX),\ \delta_j,\ T(AS,AT,ED),\ HS].$$

CS can be estimated as a function of its arguments, and the VSL can be calculated by dividing the CS by the respective level of risk. It should be noted that these two methods generally produce different estimates of the VSL.

Conclusions

Although the general models and data are available to link together the chemical fate and transport, economic, and human health models, an application of this framework may be difficult because of a lack of chemical-specific and site-specific data. Site-specific data needed for the chemical simulation models is available for only certain areas in the U.S., hence the reliance on site-specific studies instead of on national data bases.

Because most management-oriented chemical fate and transport models do not simulate chemical movement down to the groundwater zone, the researcher may be forced to make assumptions about the fate of a chemical as it moves between the root zone and the groundwater zone; this could result in under- or over-estimation of potential groundwater loading.

Although using a CV method to value human health risks is consistent with the concept of potential Pareto improvement, it can be subject to a variety of biases. This problem may be further exacerbated by a lack of data needed for the explanation of the CV scenario. In particular, data that indicate levels of health risks associated with level of groundwater pollution may not be available.

CHAPTER 9

CONCLUSIONS AND RECOMMENDATIONS

This study has identified the major components of a framework for the measurement and valuation of the social costs of groundwater pollution. In general terms, the major components of this framework are currently available in the relevant disciplines. There are, however, some important gaps in methods and data. Some of these gaps are due to the multi-disciplinary nature of the problem. A multi-disciplinary approach is needed, yet most physical and economic models are designed to be used by researchers within the respective discipline. To close these gaps, researchers must consider the broad implications and potential applications of their research, both within and outside their fields. Some research recommendations necessary to close these gaps are discussed below.

A major gap in this linkage of models occurs because most management-oriented chemical fate and transport models do not simulate potential groundwater concentration resulting from a chemical application. Modeling chemical movement down into the groundwater zone is a considerable task because much research is needed on the physical, biological, and chemical processes occurring in the unsaturated and saturated zones. However, it is a task that must be completed in order to model chemical transport down to a zone that would be meaningful to

participants of a contingent valuation exercise (e.g., the zone that supplies drinking water).

One potential hindrance to the analysis of human health risks resulting from exposure to agricultural chemicals is difficulty in establishing causality. This problem of establishing causality is compounded in the case of groundwater pollution because exposure to contaminated groundwater is less direct than other means of exposure to agricultural chemicals. For example, the source of poisoning resulting from applying chemicals with a backpack sprayer may be easier to pinpoint than poisoning resulting from exposure to contaminated groundwater. Also, many of the health risks presented by agricultural chemicals are chronic effects with a long period of latency; because of the time between exposure to the chemical and manifestation of the disease, determining the cause of the disease may be difficult.

Much of the human health risk analysis currently relies on animal toxicity data or epidemiologic studies; because of inter-species differences or lack of an adequate control group, extrapolation of human risks from these types of studies may be questionable. However, as discussed in Chapter 6, Gough (1989a) presents evidence that suggests that estimating human health risks from epidemiologic and toxicologic data may be appropriate. Also as discussed in Chapter 6, the current mathematical dose-response models do not incorporate chemical-specific data; incorporating this type of information may make extrapolation from these mathematical models more reliable.

Another problem encountered when attempting to determine the potential health risks of an agricultural chemical is present because

testing of the inert ingredients of an agricultural chemical for acute and chronic health effects is not required. A complete analysis of the health effects of a chemical would include the health effects of the inert ingredients.

Although a substantial amount of research has investigated the health risks of pesticides, relatively little research has been conducted on the health risks of fertilizers. In some areas nitrate contamination of groundwater may be more of a problem than pesticide contamination; concern over the health risks of pesticides may be overshadowing the potentially greater health risks presented by fertilizers. This may be because more is known about the health risks of pesticides due to the availability of data on acute human toxicity of pesticides (e.g., especially resulting from pesticide application).

One shortcoming of the currently available data is the lack of data on location-specific environmental characteristics, such as the type of data needed for the chemical fate and transport models. Also, data on the distribution of these environmental characteristics are needed to evaluate policy impacts. Although agricultural policies may be uniformly applied throughout the nation, the impacts of the policy may not be uniform. As illustrated by Antle and Just (1989), the relationship between, and distribution of, productivity-related characteristics and damage-producing characteristics must be determined in order to estimate the effects of a policy on the environmental damage of a particular region.

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APPENDIX

APPENDIX

This appendix provides an overview of the PRZM model. Mulkey et al. (1986) present the mathematical advection-dispersion equations used in PRZM to estimate the changes over time of the chemical concentration in the compartments of the soil profile. The equation variables are:

C_w = dissolved pesticide concentration in soil water, M/L^3

C_s = sorbed pesticide concentration, M/M

θ = soil volumetric water content, L^3/L^3

V = compartment volume, L^3

ρ = soil bulk density, M/L^3

D = coefficient of dispersion and diffusion, L^2/T

v = water pore velocity, L/T

k_u = pesticide plant uptake efficiency, $0 \leq k_u \leq 1.0$

T = volumetric plant transpiration rate, L^3/T

RO = volumetric rate of surface water runoff, L^3/T

Σk_w = summation of all first-order rate constants for transformation of dissolved pesticides, 1/T

F_d = fraction of applied pesticide in dissolved form

J_{ap} = pesticide application rate, M/T

t = time, T

x = vertical distance, L

DPR = depth of surface runoff zone, L

ERO = surface soil erosion rate, M/T

Σk_s = summation of all first-order rate constants for transformation of sorbed pesticides, 1/T

where L is a one-dimensional measure of space, T is a measure of time, and M is a measure of mass.

For the surface zone where $0 \leq x \leq DPR$, the equations are:

$$\frac{\partial(C_w\theta V)}{\partial t} = \frac{\partial}{\partial x} D \frac{\partial(C_w\theta V)}{\partial x} - \frac{\partial(vC_w\theta V)}{\partial x} - k_u TC_w - ROC_w - C_w\theta V\Sigma k_w + F_d J_{ap} \quad , \text{ for } 0 \leq x \leq DPR \qquad [A.1]$$

$$\frac{\partial(C_s\rho V)}{\partial t} = - EROC_s - C_s\rho V\Sigma k_s + (1-F_d)J_{ap} \qquad [A.2]$$

Equation [A.1] determines the change in the amount of dissolved chemical in the surface zone. The processes modeled include dispersion and diffusion of dissolved chemical, advection, plant uptake of the dissolved chemical, surface water runoff of dissolved chemical, transformation processes acting on the dissolved chemical, and the amount of dissolved chemical applied to the area. Equation [A.2] determines the change in the amount of sorbed chemical in the surface zone. The processes modeled include erosion of sorbed chemical, transformation processes acting on the sorbed chemical, and the application of undissolved (solid) chemical. According to Leonard et al. (1987), PRZM is somewhat unresponsive to rainfall for surface runoff and erosion; therefore, the amount of chemical carried off the field may be understated, causing the model to estimate "worst case" leaching potential.

The equations representing the changes in the lower subsurface zones ($x \geq DPR$), differ from those of the surface zone because chemical applications and runoff are not modeled in the lower subsurface zones:

$$\frac{\partial(C_w \theta V)}{\partial t} = \frac{\partial}{\partial x} D \frac{\partial(C_w \theta V)}{\partial x} - \frac{\partial(v C_w \theta V)}{\partial x} - k_u T C_w - \Sigma k_w C_w \theta V$$

$$\frac{\partial(C_s \rho V)}{\partial t} = - \Sigma k_s C_s \rho V, \quad \text{for } X \geq DPR$$

Water Balance Equations

It is necessary to estimate the movement of water through the soil because water transports dissolved chemical. Water balance equations are separately developed for: (1) the surface zone, (2) horizons comprising the active root zones, and (3) the remaining lower horizons within the unsaturated zone. Depending on the zone, these water balance equations may be functions of precipitation, snow melt, evaporation, transpiration, percolation, and soil water contained in the soil layers. Carsel et al. (1985) present the water balance equations:

(1) for the surface zone:

$$SW_{1,t+1} = SW_{1,t} + P_t + SM_t - INT_t - R_t - E_{1,t} - PER_{1,t}$$

(2) for the root zones:

$$SW_{i,t+1} = SW_{i,t} + PER_{1,t} - E_{i,t} - T_i - PER_{i-t}$$

(3) for the lower horizon zone:

$$SW_{i,t+1} = SW_{i,t} + PER_{i-t} - PER_{i,t}$$

where

$SW_{i,t}$ = soil water in layer i for day t (cm)

INT_t = interception of precipitation by plants for day t (cm)

E = evaporation for day t (cm)

P = precipitation as rainfall for day t (cm)

R = surface runoff for day t (cm)

SM = snowmelt for day t (cm)

T = transpiration for day t (cm)

PER_i = percolation of soil water from zone i for day t (cm).

A modification of the SCSCN technique is used to distribute daily rainfall between runoff and infiltration. According to Carsel et al. (1984), this SCSCN method was chosen because it is a reliable procedure used for many years and the required inputs are generally available. The SCSCNs are determined on a daily basis.

The daily evapotranspiration demand is divided among evaporation from plant canopy, soil evaporation, and plant transpiration. Total demand is first estimated and then extracted sequentially from crop canopy storage and from each layer until wilting point is reached in each layer or until total demand is met. (The moisture content in a soil layer at which plant survival cannot be achieved and the plant permanently wilts is called the wilting point.)

The percolation component of the water balance equation can be calculated using either of two options. With Option 1, percolation is defined in terms of field capacity and wilting point. [Bear and Verruijt (1987) define field capacity as the amount of moisture remaining in the soil after an extended period of gravity drainage without an additional supply of water at the soil surface.] This option is based on the assumption that any soil water in excess of field capacity will percolate or drain freely into the next (lower) layer. Option 2 is provided to accommodate soils having low permeability layers that restrict the "free drainage" assumed in Option 1.

Erosion Equations

To calculate the amount of sorbed pesticide carried away on eroded sediments, an estimate of soil erosion is required. Soil loss is estimated using a modified Universal Soil Loss Equation (Williams, 1977) that is expressed as (Carsel et al., 1984):

$$X_e = a\ (V_r q_p)^{0.56}\ KLSCP$$

where

V_r = volume of event (daily) runoff (m^3)

q_p = peak storm runoff (m^3/sec)

K = soil erodibility factor

LS = length-slope factor

C = soil cover factor

P = conservation practice factor

a = units conversion factor.

Chemical Transport Equations

To produce soil water and solid phase pesticide concentrations (C_w and C_s, respectively), the chemical transport component calculations include pesticide uptake by plants, erosion and surface runoff losses, decay, vertical movement, foliar loss, and dispersion. Plant uptake of pesticides is modeled by assuming uptake is directly related to the plant's transpiration rate. Degradation of the pesticide is represented as a first-order process; this representation includes all significant biochemical transformation and decay processes such as hydrolysis

(chemical decomposition in the presence of water), photolysis (chemical decomposition in the presence of light), and microbial decay. The pesticide application rate is a simple input rate, but must be partitioned between the plant canopy and the soil surface.