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Models for Analyzing Agricultural Nonpoint-Source Pollution

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MODELS FOR ANALYZING AGRICULTURAL NONPOINT-SOURCE POLLUTION

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FOREWORD

The International Institute for Applied Systems Analysis is conducting research on the environmental problems of agriculture. One of the objectives of this research is to evaluate the existing mathematical models describing the interactions between agriculture and the environment.

Part of the work toward this objective has been led at IIASA by G.N. Golubev and part has involved collaboration with several other institutions and scientists. During the past two years the work has paid particular attention to the problems of pollution from nonpoint sources.

This report reviews and classifies the mathematical models currently available in this field, taking into account their different time and spatial scales, as well as the problems that may call for their use.

Although the last decade has witnessed the rapid development of nonpoint-source pollution models, much remains to be done. Haith addresses this matter; however, his comments about research needs go beyond the art and science of modeling.

JANUSZ KINDLER
Chairman
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SUMMARY

Mathematical models are useful means of analyzing agricultural nonpoint-source pollution. This review summarizes and classifies many of the available chemical transport and planning and management models. Chemical transport models provide estimates of chemical losses from croplands to water bodies; they include continuous simulation, discrete simulation, and functional models. A limited number of transport models have been validated in field studies, but none has been tested extensively. Planning and management models, including regional impact, watershed planning, and farm management models, are used to evaluate tradeoffs between environmental and agricultural production objectives. Although these models are in principle the most useful for policy-making, their economic components are much better developed than components for predicting water pollution.

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1 INTRODUCTION

The management of agroecosystems is usually for productive purposes. Land resources are subjected to meteorological inputs and management practices to yield desired biological outputs of food and fiber. The "desired" outputs and necessary management practices are determined by policy decisions of national and regional authorities and farm operators. These decisions may be mixtures of tradition, rational planning, and responses to economic stimuli. Regardless of their origin, however, agricultural policies are shaped primarily by their perceived effects on food and fiber production.

Twentieth century agricultural planners have learned that chemical inputs to crop production, in the form of fertilizers and pesticides, can be highly efficient means of increasing yields. In additon, the control of water inputs through irrigation has become a major factor in the conversion of arid regions to productive farmlands. Unfortunately, the agricultural policies which have encouraged irrigation and chemical use have not only increased efficiency, but have also produced distributions of chemical residuals in the environment that have degraded water quality. These water pollution impacts are largely unintentional. On nonirrigated land they are associated with diffuse or nonpoint sources that are caused by natural hydrologic phenomena. With irrigated agriculture, nonpoint-source pollution is often caused by return flows that carry the leaching waters necessary to maintain favorable salt balances for crop growth.

When the water quality problems caused by agricultural nonpoint sources become severe, production practices may need to be evaluated for both their economic and environmental consequences. As the control of agricultural pollution has relatively recent emphasis even in developed countries, past experience provides little assistance, and it has been necessary to rely on mathematical models as tools for policy evaluation.

Models have been developed for two major purposes. The first is the estimation of the water pollution impacts of agricultural production and pollution control practices. The second is the analysis of tradeoffs between agricultural production and environmental quality objectives.

A large number of nonpoint-source models have been constructed and are now available for agricultural and water quality planners. These models vary significantly in structure, underlying assumptions, and purpose. This diversity is due larely to the pressing need to resolve policy issues related to agricultural pollution. Modeling research has often been problem-oriented, and there has been little time for the long-term investigations that are necessary for the orderly development of scientific theory. Rather, engineers and scientists from different disciplines responded to urgent needs with models which are capable of providing some of the more critical information required for rational policy-making.

This report is a review of these first-generation agricultural nonpoint-source models and has two broad objectives: (1) to organize the immense variety of models into a framework, or system of classification which can usefully highlight significant model differences and similarities; and (2) to summarize model characteristics which are likely to be of interest to potential users; i.e., to provide a catalogue or a user's guide to the state-of-the-art. The review is largely descriptive and does not critically evaluate the mathematical characteristics of the models; however, it attempts to provide a current assessment of modeling directions.

The report is divided into three sections. The first is devoted to chemical transport models. These are models designed to predict the losses of salts, nutrients, and pesticides from agricultural lands. Such models can in principle be linked to water quality models which estimate the effects of transported chemicals on water quality. Water quality models are not unique to nonpoint sources since they are in general designed to predict the response of a water body to both point and nonpoint sources. The literature contains many examples of such models and they are omitted from this review. Sediment transport models are also omitted, partly in the interest of brevity, but also as a reflection of the fact that sediment *per se* is seldom a critical or manageable water quality problem. Sediment is important mainly as a carrier of chemicals, and sediment models are integral components of many chemical transport models. The second section of the report is devoted to planning and management models for agricultural pollution. Most of these are linear programming models which are used to analyze the environmental and economic impacts of nonpoint-source controls. The final section concludes and suggests possible directions for future modeling research.

2 CHEMICAL TRANSPORT MODELS

The major hydrologic processes which transport chemicals from cropland to surface or groundwater bodies are shown in Figure 1. Omitted from the figure are atmospheric interactions whereby volatilized chemicals or aerosols are transported to surface waters. The significance of such air-borne pollution is largely unknown, and there have been few attempts to model these phenomena. The hydraulic components of nonpoint-source pollution are surface runoff, subsurface runoff (interflow), and percolation. The latter two flows can transport dissolved chemicals while surface runoff may carry both dissolved and solid-phase (particulate) chemicals. Solid-phase chemicals travel with sediment that has been eroded from the land surface and carried by surface runoff. Transport models may be designed to predict losses of chemicals from the land surface and soil in one or more of the possible water components. Relatively few models are capable of complete description of all of the transport pathways.

2.1 Model Types and Characteristics

There are obviously many different ways of classifying a subject as broad and fragmented as nonpoint-source models, and the system proposed here is preliminary and somewhat arbitrary. In general, the system was designed to capture the significant differences and similarities among models and provide summary information to potential users. In addition, the method of classification was constrained by the need to accommodate the 37 widely varying chemical transport models which are included. The models are described by six general characteristics:

1. Model Structure Type
2. Principal Outputs
3. Scale
4. Time Step
5. Calibration
6. Validation Studies

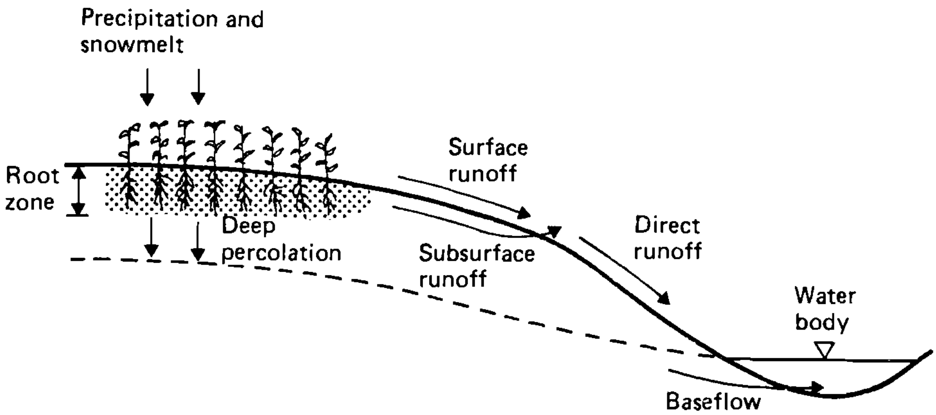

FIGURE 1 Transport processes for nonpoint-source pollution.

2.1.1 Model Structure Type

There are different types of chemical transport models. The first, and most analytical, are continuous simulation models. These models are based on either systems of partial differential equations for water and solute transport or on kinetic models described by ordinary differential equations. The second are discrete simulation models. Such models are sets of algebraic equations which describe discrete changes over time. Because the models are solved algebraically they are typically easier to manipulate than continuous models. The simplest transport models, which can be classified as functional models, differ from simulation models in that they seldom attempt to capture the details of the actual biological, chemical, or physical processes which affect chemical losses. Rather, they are simple equations which predict chemical losses based on intuitive or empirical information.

2.1.2 Principal Outputs

This model characteristic is largely self-explanatory, and accounts for many of the significant differences in models. Models are described by both the chemicals they portray – salts, nitrogen (N), phosphorus (P), or pesticides – and the hydraulic distribution of chemical losses – surface runoff, subsurface runoff, percolation.

2.1.3 Scale

Scale refers to assumptions of spatial homogeneity. Field models assume that the soil surface is horizontally homogeneous, thus they are applicable to a single "field" with a uniform soil type. Watershed models can be used to describe heterogeneous drainage areas, and in particular the distribution of chemical sources from different fields and their aggregation for an entire watershed.

2.1.4 Time Step

Model time step is an important characteristic for potential users, since it is an indicator of computational and meteorologic data requirements. Model computations must be repeated for each time step, and hence models with small time steps are often more costly to use.

2.1.5 Calibration

Calibration involves the use of a model to estimate its own parameters. In general, a model must be calibrated if, in applying the model to a specific physical setting (field or watershed) it is necessary to measure phenomena which the model is designed to predict. The purpose of the measurements is to provide values for model parameters which would otherwise be difficult, if not impossible to estimate. Calibration is a complex issue in nonpoint-source modeling and involves both practical and philosophical considerations which are both fundamental and somewhat subjective.

The process of calibration can be considered a rational response to uncertainty. No transport model for agricultural chemicals can be more than a crude approximation of reality. By providing for calibration, the modeler can include mathematical descriptions of processes whose parameters defy simple evaluation based on commonly available soil, crop, or chemical properties. In addition, by calibrating a model to a monitored situation, greater predictive accuracy may be obtained. Although this argument is in principle correct, it must be recognized that the calibration process may mask model limitations. When the physical and chemical processes within a model are described by analytical relationships based on generally accepted scientific theory, the adjustment of several parameters by calibration may be a sound procedure. Unfortunately, calibration parameters sometimes do not correspond either to rational analytical relationships or recognizable physical or chemical properties of the transport processes. In this case, calibration may be an arbitrary scaling of model predictions to force an otherwise inadequate model to yield reasonable results.

Most calibration needs fall somewhere between the two extremes, and the classification system used in this report does not attempt to evaluate the degree to which a model may be compromised by calibration. To some extent, any such assessment would be subjective. However, it is apparent that any need for calibration imposes constraints on a model's general applicability. Agricultural nonpoint-source models must be used ultimately to evaluate management practices, and one can seldom guarantee that changes in management from a calibrated situation will not change the calibrated parameters. Furthermore, since models requiring calibration cannot be applied to unmonitored sites, they are of limited usefulness in studies where resources do not permit such monitoring.

As a final point, it should be noted that in spite of the problems caused for potential users, a model's need for calibration is not necessarily a negative attribute. A calibrated model may provide a more realistic description of chemical transport than an alternative model which has no calibration parameters. Difficulties in measurement of parameters may not imply that a model is unscientific. In addition, increased experience in applying a model may lead to simpler means of parameter estimation. In this fashion, experience may eliminate the calibration requirement.

2.1.6 Validation Studies

A complete discussion of model validation is well beyond the scope of this report, and in the present context this classification category refers only to whether or not there has been a documented attempt to determine the accuracy of a model's predictions by comparison with measured chemical transport losses. Such an evaluation must be at the intended scale of the model (field or watershed rather than laboratory) and be based on different measurements than those used for calibration. Given the unavoidable errors in the collection and analysis of chemical losses from croplands and uncertainties in model parameter estimates, it is difficult to see how any transport model can ever be shown to be "valid." Thus, the comparison of model predictions with observations is largely subjective. Nevertheless, these comparisons provide the only quantitative indicator of the validity of a model as an abstraction of reality. Many chemical transport models have not been subjected to such testing and hence are not yet suitable as general tools for either estimating agricultural pollution or evaluating management practices.

2.2 Continuous Simulation Models

Chacteristics of 13 simulation models are listed in Table 1. Model time steps are not provided since all the models are based on differential equations and can be solved analytically or numerically for arbitrary time increments. With two exceptions (Amberger et al., 1974; Konikov and Bredehoeft, 1974), all the models are limited to percolation losses from a field and/or groundwater transport in a watershed (aquifer).

TABLE 1 Continuous simulation models for agricultural chemical transport.

Reference	Principal output	Scale	Calibration required?	Validation studies?	Model structure
Davidson et al. (1978) ("Research Model")	N in percolation	Field	Yes	No	One-dimensional D'Arcy flow and convection/dispersion partial differential eqns.
Davidson et al. (1978) ("Management Model")	N in percolation	Field	No	Yes	"Piston displacement" of water, 1st order kinetics for N
van Veen (1977)	N in percolation	Field	Yes	Yes	No water model, ordinary differential eqns. solved by CSMP
Mishra et al. (1979)	P in percolation	Field	Yes	Yes	No water model. 1st order kinetic eqns. solved by CSMP
Shah et al. (1975)	P in percolation	Field	Yes	Yes	One-dimensional D'Arcy flow and convection/dispersion partial differential eqns.
Mansell et al. (1977b)	P in percolation	Field	Yes	No	One-dimensional convection/dispersion partial differential eqns. No water model.
Mansell et al. (1977a)	P in percolation	Field	Yes	No	One-dimensional convection/dispersion partial differential eqns. Kinetic models for exchange of P forms. No water model.
O'Connor et al. (1976	Pesticide in percolation	Field	Yes	No	One-dimensional convection/dispersion partial differential eqns. No water model.
Davidson et al. (1975	Pesticide in percolation	Field	Yes	Yes	One-dimensional D'Arcy flow and convection/dispersion partial differential eqns.

Amberger et al. (1974)	N in percolation, P in surface runoff	Field	Yes	No	Differential eqns. for water and N movement. Erosion modeled and all P losses assumed solid-phase. Solution by CSMP. No model testing.
Czyzewski et al. (1980)	N in percolation and groundwater	Watershed	Yes	No	Two-dimensional D'Arcy flow and convection/dispersion partial differential eqns. No model testing.
Konikow and Bredehoeft (1974)	Salts in return flows, river and groundwater	Watershed	Yes	Yes	Conjunctive aquifer and river model. Two-dimensional. D'Arcy flow and convection/dispersion partial differential eqns. for groundwater, simple mass balance for river.
Mercado (1976)	N, salts in percolation and groundwater	Watershed	Yes	No	Treats aquifer as single cell with complete mixing. Analytical solution to 1st order equation.

Ten of the models are field-scale models designed to predict vertical movement of soil chemicals in percolation waters. Six of the ten models are based on the general convection/dispersion equation for transport of a reactive solute in a porous medium and are a sample of many comparable models that have appeared in the literature. The "research model" of Davidson et al. (1978) is the most complete of these models, providing detailed analytical descriptions of N sources, sinks, and transformations as well as a complete water balance. The model is difficult to solve and is very data intensive. Although the three models developed by Shah et al. (1975) and Mansell et al. (1977a, 1977b) are designed for similar purposes, only the first incorporates a water flow component and has been subjected to validation studies. Similarly, the two pesticide models (O'Connor et al. 1976; Davidson et al. 1975) are both designed for estimating percolation losses, but only the latter includes a water model and has been tested in validation studies.

As a generalization, models for chemical losses that are based on convection/dispersion equations must be calibrated and are not easily verified. When such models incorporate water balances they are difficult to solve for realistic boundary conditions. The rationale for this modeling approach has been that it is a fundamental and hence realistic theory for chemical movement through the soil. This view has been challenged by Sposito et al. (1979):

> . . . none of the existing foundation theories has yet achieved the objectives of: (1) deriving, in a physically meaningful and mathematically rigorous fashion, the macroscopic differential equations of solute transport theory, and (2) elucidating the structure of the empirical coefficients appearing in these equations.

However, these same general objections are applicable to any chemical transport model, and they are not sufficient reasons for rejecting the convection/dispersion approach.

Three of the continuous simulation models have structures somewhat similar to the discrete simulation models (Section 2.3). However, they are based on differential equations and are solved by the IBM Continuous System Modelling Program (CSMP). The van Veen (1977) and Amberger et al. (1974) models provide very detailed descriptions of soil N processes. Both models must be considered preliminary, since the former has yet to incorporate plant uptake of N and soil moisture balances and the latter has not been tested at any scale. The model developed by Mishra et al. (1979) for P transformations in forest soils is the most operational of the CSMP models since it is both relatively simple in structure and has been tested with validation studies.

The "management model" of Davidson et al. (1978) is a simplified version of their "research model" and provides a very straightforward means of estimating percolation losses of N. This model, which has been validated, is the only continuous simulation model that does not require calibration. Of all the models listed in Table 1, it is probably the only one which is currently suitable for a general user.

Two of the watershed models (Czyzewski et al., 1980; Konikow and Bredehoeft, 1974) are attempts to describe chemical distributions in aquifers. The Czyzewski model is intended for application to a large portion of the Skrwa River Basin in Poland. The model is preliminary at this time, and major programs of data collection and testing will be necessary to make it operational. Konikow and Bredehoeft's model links surface and groundwater flows and has been successfully applied to a portion of the Arkansas River in Colorado.

2.3 Discrete Simulation Models

The 19 simulation models listed in Table 2 fall into three groupings: percolation models, models based on complete hydrologic balances, and models for irrigation return flows.

2.3.1 Percolation Models

The first seven models are designed to estimate percolation losses of dissolved N from fields. One of the models (Dutt et al., 1972) is also capable of estimating salt losses. The models developed by Addiscott (1977), Haith (1973), and Saxton et al. (1977) are similar in that they are restricted to situations where runoff is either negligible or is provided as model input. Each of these models is based on relatively simple N balances and has modest data and computational requirements. Addiscott's model is the only one of the three that does not require calibration, although it has only been validated for nongrowing season conditions. The next two models (Duffy et al., 1975 and Tanji et al., 1979) are heavily empirical and have not been validated with data sets other than those used for calibration. Since both models require adjustment of many calibration parameters, they do not appear suitable for general use.

The final two percolation models are somewhat unique. The model for percolation N losses given in Stewart et al. (1976) has a complete hydrologic balance component including runoff, although it does not predict losses of N in runoff. This type of hydrologic model, based on the US Soil Conservation Service's (SCS) runoff equation, is similar to several models discussed in the next model group. The model does not require calibration, but has not been validated. The model of Dutt et al. (1972) was one of the first agricultural transport models. It is in many ways a hybrid, since it has a water flow component similar to the continuous simulation models. The time step of 0.1 da is somewhat misleading since portions of the model require iterative computations at much greater frequencies. In spite of its precedence over later models, it does not appear to have seen significant use, probably due to its extensive computational and data requirement.

2.3.2 Complete Hydrologic Models

Nine of the remaining models contain complete hydrologic budgets. Three models Frere et al., 1976; Tseng, 1979; Williams and Hann, 1978) are designed to estimate watershed chemical export in streamflow, and the latter two have been incorporated in watershed planning models. Watershed models differ from field models in that the former consider the variations in soils and crops in a large drainage area and integrate distributed chemical losses into a time series of total chemical mass fluxes from the watershed. Such an integration is extremely difficult and it is not surprising that only the simplest of the three models (Tseng, 1979) has been validated. The model of Williams and Hann is the most complete watershed model, although it does not include dissolved P losses.

The first four field-scale models (Haith, 1979; Knisel, 1980; Haith, 1980; Steenhuis, 1979) have similar hydrologic structure based on the SCS runoff equation. However, the Knisel and Steenhuis models have options permiting infiltration calculations based on the Green and Ampt infiltration equation at hourly time steps. The Cornell Nutrient Simulation (CNS) model (Haith, 1979) is a relatively efficient model that does not require calibration. Daily water balances are aggregated for the monthly nutrient submodel. The US

TABLE 2 Discrete simulation models for agricultural chemical transport.

Reference	Principal outputs	Scale	Time step	Calibration required?	Validation studies?	Model structure
Addiscott (1977)	N in percolation	Field	da	No	Yes	Separation of soil water into mobile and retained fractions. No runoff.
Haith (1973)	N in percolation	Field	mo	Yes	Yes	Analytical eqns. for mineralization and leaching. Regression eqns. for crop up-take. No runoff.
Saxton et al. (1977)	N in percolation	Field	da	Yes	Yes	Simple N model combined with more complex soil moisture model. No runoff.
Duffy et al. (1975)	N in percolation (tile flow)	Field	12 hr	Yes	No	Empirical eqns. for runoff and most N process in soil.
Tanji et al. (1979)	N in percolation	Field	yr	Yes	No	Water and N processes described by empirical linear eqns.
Stewart et al. (1976)	N in percolation	Field	da	No	No	Piston flow with exclusion factor. Nitrification is major N process. SCS runoff eqn.
Dutt et al. (1972)	N, salts in percolation	Field	$\leqslant$ 0.1 da	Yes	Yes (N)	D'Arcy one-dimensional water flow. N reaction rates based on regressions. No runoff.
Frere et at. (1975)	N, P, pesticde in streamflow	Watershed	hr	Yes	No	Based on USDA runoff model (Holtan et al., 1975).
Tseng (1979)	N in streamflow	Watershed	hr	Yes	Yes	Dissolved N. Based on USDA runoff model (Holtan et al., 1975). Watershed divided into hydrologic zones.

Williams and Hann (1978)	N, P in streamflow	Watershed	da	Yes	No	Solid-phase P, dissolved and solid-phase N. SCS runoff, USLE erosion. Stream routing functions, synthetic hydrographs.
Haith (1979)	N in total runoff and percolation, P in runoff	Field	da	No	Yes	Solid-phase and dissolved chemicals, SCS runoff USLE erosion. Synthetic hydrograph (peak flow).
Knisel (1980)	N, P, pesticide in total runoff and percolation	Field	da	No	No	Solid-phase and dissolved chemicals. SCS runoff eqn., synthetic hydrograph (peak flow). Multiparameter erosion/sedimentation model.
Haith (1980)	Pesticide in total runoff	Field	da	No	Yes	Dissolved and solid-phase pesticides. SCS runoff, USLE erosion.
Steenhuis (1979)	Pesticide in total runoff, percolation	Field	da	Yes	Yes	Dissolved and solid-phase pesticides. SCS runoff, USLE erosion.
Donigian et al. (1977)	N, P, pesticide in surface, subsurface runoff, base flow	Field	5–30 min	Yes	No	Kinetic models for soil chemicals. Stanford Watershed model for moisture balance, analytical erosion model.
Bruce et al. (1975)	Pesticide in runoff	Field	5 min	Yes	No	Empirical characteristic functions for runoff, sediment transport. Pesticide concentrations by regression. No pesticide decay.
Riley and Jurinak (1979)	Salts in return flows	Watershed	Steadystate	Yes	No	Simple salt balances based on measured river salinity.
Scherer (1977)	Salts in return flows and river	Watershed	Steadystate	No	No	Simple mass balances for water, salt in soils and river. No groundwater interactions.
Bardaie (1979)	Salts in return flows and river	Watershed	yr	Yes	Yes	Extension of Scherer (1977) model to predict changes over time in soil and river salinity.

Department of Agriculture CREAMS (Chemicals Runoff and Erosion from Agricultural Management Systems) model (Knisel, 1980) has many structural similarities to the CNS model and differs chiefly in its handling of erosion and sediment transport. The CREAMS model includes sediment detachment, transport, and deposition based on particle size distribution, while the CNS model estimates sediment losses by event-based modifications of the Universal Soil Loss Equation (USLE). The CREAMS model, which has not yet been validated, can in principle be used without calibration although many of its parameters, particularly those for sediment transport, are very difficult to estimate. The two pesticide models (Haith, 1980; Steenhuis, 1979) are similar in structure, but the Steenhuis model is unique in its ability to estimate the downward movement of pesticides in the soil.

Although the Agricultural Runoff Management (ARM) model developed by Donigian et al. (1977) produces output similar to the CREAMS model, it has very different hydrologic and sediment components. The model's foundation is the Stanford Watershed Model that determines outflow hydrographs from catchments based on a calibration approach for infiltration, subsurface runoff, and soil moisture capacities.

The final model in this category (Bruce et al., 1975) is completely empirical. It is designed to estimate pesticide losses during runoff events. It does not consider the dynamics of pesticide decay between events, and hence does not have the capabilities of the other pesticide models.

2.3.3 Irrigation Return Flow Models

There are a variety of models designed to analyze salinity problems for irrigated agriculture (see for example, the review by Walker, 1977). The three listed in Table 2 (Riley and Jurinak, 1979; Scherer, 1977; Bardaie, 1979; also described in Bardaie and Haith, 1979) are not necessarily typical, but unlike many other models, they are designed to evaluate both the magnitudes of salt fluxes in return flows and their effects on downstream diversions. Salinity models differ significantly from other nonpoint-source models in that they are concerned with conservative chemicals and well-defined drainage systems to transport leached chemicals to surface waters. Runoff prediction is usually not important, and model structures are based on simple mass balances for water and salinity.

2.4 Functional Models

The advantages and disadvantages of functional models for prediciton of chemical transport are relatively apparent. Functional models are useful since they provide answers with minimal computational effort and data requirements. As such, they have been important tools in providing the preliminary estimates of chemical losses needed to complete many of the early studies of agricultural nonpoint-source pollution. Unfortunately these advantages are mostly operational. Since functional models do not attempt to simulate the fundamentals of chemical transport processes, they may not be reliable bases for designing pollution control programs.

Characteristics of five functional models for chemical transport are given in Table 3. The Burns (1974, 1975) N percolation model is the simplest and perhaps most reliable of the models. It consists of a simple leaching equation that is capable of predicting the

TABLE 3 Functional models for agricultural chemical transport.

Reference	Principal outputs	Scale	Time step	Calibration required?	Validation studies?	Model structure
Burns (1974, 1975)	N in percolation	Field	Arbitrary	No	Yes	Downward movement of nitrate during nonplowing season. No water model.
Haith and Tubbs (1980)	N, P in rotal runoff	Field/ watershed	da	No	Yes	Runoff from SCS equation multiplied by dissolved concentrations, erosion from USLE multipled by soil concentrations.
Bogardi and Duckstein (1978)	P in surface runoff	Field/ watershed	da	Yes	No	Dissolved losses based on SCS runoff, solid phase losses from USLE.
McElroy et al. (1976)	N, P, pesticides in surface runoff	Field/ watershed	Steady state	No	No	Losses based on USLE erosion multiplied by concentrations. Watershed sediment delivery from drainage density.
Holy et al. (1980)	N, P in surface runoff	Field	Arbitrary	Yes	No	Runoff from free surface flow partial differential eqns. Sediment, nutrient fluxes from regressions.

downward displacement of N (nitrate) in the soil profile. The quantity of N available for movement and percolation volume must be known. Nitrogen sinks and sources are not considered explicitly. Haith and Tubbs (1980) have tested a functional model based on the SCS runoff and USLE equations. When applied to a watershed, nutrient losses are computed from each field and summed for estimates of watershed export. The model of Bogardi and Duckstein (1978) is similar, but is limited to phosphorus and requires calibration. Both models are event based; i.e., they compute losses for each runoff event.

The "loading functions" proposed by McElroy et al. (1976) are based on average annual sediment losses predicted by the USLE. Although these functions are reasonable only for solid-phase chemical losses and have not been validated, they have been widely used. Watershed losses are determined by multiplying aggregated field losses by sediment delivery ratio. The final model, proposed by Holy et al. (1980), is a hybrid. It contains a continuous runoff model consisting of the general partial differential equations for free surface flow. Conversely, nutrient and sediment fluxes in runoff are determined by regression equations. This model has yet to be tested, and the contrasting levels of detail in the runoff and nutrient components result in greater data and computational requirements than other functional models.

3 PLANNING AND MANAGEMENT MODELS FOR AGRICULTURAL NONPOINT SOURCES

Planning and management models are designed to analyze the economic implications of alternative policies or management practices for controlling agricultural nonpoint sources. This type of analysis is necessary for evaluation of tradeoffs between environmental and production objectives. Models are important because agricultural systems are usually too complicated for the impacts of environmental control policies to be readily apparent. Furthermore, the maintenance of agricultural productivity and/or income are usually of such importance that policy-makers are reluctant to implement new regulatory programs without documentation of economic impacts.

3.1 General Approach

Unlike chemical transport models, planning and management models are all basically similar. They are based on a budgeting approach which quantifies resource requirements, financial benefits and costs, and other relationships between agricultural management activities. Budgeting is frequently within the context of optimization and most planning and management models are solved by linear programming (LP). The different types of studies can be illustrated by the general LP model:

$$\text{Max/Min } Z = \bar{c}\,\bar{X} \tag{1}$$

$$\bar{\mathrm{A}}\,\bar{\mathrm{X}} = \bar{b} \tag{2}$$

$$\bar{\mathrm{X}} \geqslant 0 \tag{3}$$

In this model $\bar{\mathbf{X}}$ is a vector of agricultural management practices which can include crop/soil combinations, chemical applications, livestock numbers, etc. Costs or returns $\bar{c}$ are asssociated with the activities, and the relationships between activities are indicated in eqn. (2), where $\bar{\mathbf{A}}$ is a matrix of activity coefficients and $\bar{b}$ a vector of resource or other physical limits.

This type of optimization model can be manipulated in several ways to explore the impacts of pollution control measures on costs or income, Z:

1. The constraint set (eqn. 2) can include budgeting of pollutant losses resulting from each activity. The associated right-hand side constants (elements of $\bar{b}$) are upper limits of total pollutant losses. These constants can be progressively tightened to determine changes in total income or costs, Z.
2. Activities can be added to or subtracted from $\bar{\mathbf{X}}$. For example, certain pesticides may be banned and new tillage practices added.
3. Characteristics of activities, which affect pollutant losses, can be changed. For example, the fertilizer application associated with a particular crop may be reduced. Such changes will modify certain of the coefficients in $\bar{\mathbf{A}}$.
4. The costs and returns associated with certain activities can be modified to reflect subsidies or taxes, offsite damages (e.g., damages to a downstream irrigator due to saline return flows), or onsite benefits (e.g., improved soil productivity with erosion control).

3.2 Characteristics of Modeling Applications

The 19 planning and management models which are summarized in this paper fall into three distinct groups. Regional impact models are designed for macroscale evaluation of the impacts of environmental and agricultural management policies on crop distributions, farm and consumer prices, and income and other aggregated economic measures. These models cover large geographic areas and usually must consider (sometimes only implicitly) supply and demand relationships. Watershed planning models are applied in the context of specific water quality problems such as reservoir eutrophication or sedimentation. The objective is to develop a comprehensive program for control of agricultural practices and point sources, if necessary, to efficiently meet water quality objectives. The third group is farm management models that are designed to evaluate the impacts of pollution control on the income and management practices of an individual farmer.

Within each group, the modeling studies are summarized with respect to five characteristics:

Environmental emphasis

The most common modeling application is to sediment control, primarily because of the availability of simple sediment models (the USLE and sediment delivery ratios) that are easily incorporated into optimization models. However, other environmental pollutants that have been studied are pesticides, nutrients, and salinity.

Location

Unlike the chemical transport models, planning and management models have little identity beyond specific applications. Hence most of the latter models have been tested in actual locations.

Optimization technique

Those models which incorporate optimization are solved by either linear programming or dynamic programming (DP).

Method for pollution estimation

In several cases, the models contain no direct estimates of pollution. More commonly, estimates are based on the USLE or simple functional chemical transport models. The most interesting and realistic models contain pollutant loss estimates based on discrete simulation models. In these situations, a two-phase modeling procedure is followed in which simulation is used to generate chemical transport data, and management programs are selected by an optimization model.

Policy implications

Planning and management models have little intrinsic value and are useful only to the extent that they provide information for policy-making. Hence this model characteristic, which summarizes the relevant information produced by the model applications, is probably the most relevant indicator of the value of a particular modeling study.

3.3 Regional Impact Models

Application of regional planning models are summarized in Table 4. The four applications are modifications of two large LP models that describe either the entire US agricultural sector or the cornbelt states. In the first of these applications (Heady and Vocke, 1979) a national model of 105 producing, 51 water supply, and 28 market regions was used to evaluate effects of restrictions on cropland erosion and N fertilizer applications. The transport of eroded soil or N to waterways was not included, so no evaluation of water pollution was made. Erosion restrictions were imposed limiting soil loss from each land type to levels that would maintain soil productivity. Nitrogen fertilizer applications were constrained to 55 kg/ha. As indicated in Table 4, although the restrictions have little national impact, regional changes can be severe, since soils in some regions are much more subject to erosion than those in other regions.

The second national application (Wade and Heady, 1978) involved a more sophisticated application of the large model used by Heady and Vocke. The model was modified to include not only erosion estimates, but also methods for transporting the eroded soil to streams and subsequent entrapment of the sediment in reservoirs. Sediment fluxes were estimated in the 18 major US river basins. Wade and Heady's model is the only one of the four models in Table 4 that is capable of directly estimating water quality impacts (sediment fluxes, in this case). Two types of constraints were investigated: restrictions on sediment fluxes in each basin and restrictions on farm level erosion similar to those in Heady and Vocke (1979). A general result of the study was that uniform controls on

TABLE 4 Application of regional planning models to agricultural nonpoint source pollution problems.

Reference	Environmental emphasis	Location	Optimization procedure	Method of pollution estimation	Policy implications
Heady and Vocke (1979)	1. Erosion 2. N fertilizer	Entire US	LP	USLE	1. Uniform restriction on soil loss and fertilizer use will have modest effects on national prices and production value. 2. Large regional shifts in crops and farm income will be produced.
Wade and Heady (1978)	River sediment fluxes	Entire US	LP	USLE. sediment delivery ratios, reservoir entrapment.	1. Minimizing cropland sediment contributions increases aggregate commodity costs 42%. 2. Erosion control at farm level produces different results (distributions of costs and sediment) than restrictions on river basin sediment loads.
Taylor and Frohberg (1977)	1. Erosion 2. Insecticides 3. Herbicides 4. N fertilizer	US cornbelt	LP	USLE	1. Extreme restrictions on chemical use (banning pesticides, reducing N fertilizer) have more adverse effects on consumers than farmers. 2. Soil loss taxes have little effect on commodity prices but decrease farm income sharply.
Seitz et al. (1979)	Erosion	US cornbelt	LP	USLE	1. Aggregate impacts of soil erosion control on farm income are small. 2. Selective erosion controls in one cornbelt state but not others affect crop distributions among states, but cause little change in farm incomes in each state.

farmland erosion are relatively expensive means of reducing river sediment loads since they are not limited to cropland that is most erosive and/or has high sediment delivery.

The two cornbelt studies were based on the same general equilibrium LP model. This model included supply and demand relationships and quantified the distribution of control costs among regions, farmers, and consumers. Taylor and Frohberg (1977) evaluated economic effects of three rather restrictive environmental policies – insecticide and herbicide bans and reductions in fertilizer N applications to 55 or 110 kg/ha. The most significant conclusion was that such policies would in general benefit farmers but increase consumer costs. The same model was subsequently modified and used by Seitz et al., (1979) in additional studies of impacts of erosion control. The aggregate costs of such controls on farm income were small, but again, consumer food prices increased under certain restrictions.

The four studies listed in Table 4 illustrate the types of broad policy implications that can be generated by planning models. In general it would appear that the primary economic impacts of agricultural pollution control policies are associated with the distribution of costs and benefits among regions, producers, and consumers. Aggregate national or regional crop production and farm income do not appear to be greatly changed by most pollution control practices.

3.4 Watershed Planning Models

Nine applications of watershed planning models are summarized in Table 5. The policy implications of each study are limited to the specific watershed that was modeled and are not necessarily generalizable to other watersheds.

The first application (Alt et al., 1979) illustrates a standard approach to analysis of watershed pollution. The specific problem addressed was sedimentation of a downstream reservoir. An LP model of the watershed's cropland was used to evaluate erosion limits, constraints on sediment flux to the reservoir and subsidies for soil and water conservation practices. Reservoir sedimentation and soil and water conservation were also the subject of the work by Reneau and Taylor (1979), but their study included a much more complete accounting of social benefits and costs. Offsite sediment damage functions based on reservoir dredging and cleaning of flood control structures were included and the productivity benefits of soil conservation were estimated. Even so, it was determined that erosion control measures could not be economically justified in the watershed.

Onishi and Swanson (1974) also included offsite dredging costs in their model, but in this case it was optimal to reduce farmland erosion. Because the study also included nitrate leaching, the relationship between two environmental problems, groundwater pollution and reservoir sedimentation, could be investigated. As might be expected, sediment (erosion) controls did not also serve to control N pollution of groundwater.

The Casler and Jacobs (1975) model is similar to that of Alt et al. (1979), since P losses from a watershed in streamflow were all assumed to be associated with eroded soil. Hence erosion was the primary process modeled and P losses were obtained by multiplication by a constant. One of the general results of this study, which was also seen in most of the other applications, is that the marginal costs of nonpoint-source pollution control increase dramatically as higher levels of pollution reduction are sought.

The watershed modeling approach of Wineman et al., (1979), which is based on a study by Meta Systems, Inc., is unique. It establishes a modeling framework designed to describe the processes of agricultural nonpoint-source pollution from their origin in a farmer's field to their ultimate water quality impacts. At the present time, the approach is preliminary since although some testing has been done on the Black Creek watershed in Indiana, the general validity of the models has not yet been demonstrated.

The last four watershed applications involve models that are based on discrete simulations of pollutant transport. Unlike most other planning and management models, these four models incorporate detailed mathematical descriptions of the processes associated with nonpoint-source pollution. Model data needs are extensive, since many pollutant and economic parameters must be provided. The simulation models used in these studies are summarized in Table 2.

Williams and Hann (1978) have developed a decision theory methodology for evaluating strategies for controlling agricultural nonpoint sources. An LP model is designed that maximizes a weighted sum of decision-makers' utilities subject to water quality constraints. Coefficients for chemical and sediment losses are provided from the simulation model. The approach appears to be computationally feasible, but no attempt was made to actually estimate utilities. Tseng's (1979) model also involves a two-step modeling approach, but in this case simulation is used as a means of evaluating the environmental effects of land use plans produced by an optimization (LP) model.

In the two salinity models (Scherer, 1977; Bardaie, 1979), the simulation models described in Table 2 are integral parts of an optimization model. In Bardaie's model, this integration produces a model that is difficult to solve, and simplifications were necessary to obtain solutions by either DP or separable LP.

3.5 Farm Management Models

The final group of models presented in this section are extensions of the standard LP farm planning and budgeting models that have been used for many years. The addition of environmental parameters to such models has been a logical means of exploring the effects of agricultural pollution control on farm management. Unlike regional and watershed models, farm models are microscale, and provide estimates of the impacts of environmental policies on the farmer's day-to-day activities. In general, the models should provide more sensitive indicators of the impacts of policies than the larger scale models are capable of.

Each of the six models listed in Table 6 provides estimates of cropland erosion using the USLE. Only one model (Smith et al., 1979) combines erosion with sediment delivery ratios to determine the losses of eroded soils to surface waterways. Thus, Smith et al. were able to compare erosion and sediment control programs. They concluded that uniform imposition of erosion controls on all of a farmer's fields is not an efficient way to control stream sediment losses. This result is significant since it suggests that policies to control sediment are not equivalent to, and may be incompatible with, other policies to reduce erosion.

White and Partenheimer (1979) modeled 12 Pennsylvania dairy farms to evaluate the effects of adopting soil conservation plans recommended by the US Soil Conservation

TABLE 5 Applications of nonpoint-source planning models to watershed water quality management.

Reference	Environmental emphasis	Location	Optimization procedure	Method of pollution estimation	Policy implications
Alt et al. (1979)	Sediment	Iowa River-Coralville Reservoir, Iowa (3,800 km²)	LP	USLE, delivery ratios	Large (75%) reductions in sediment flux possible with small (4%) increase in farm costs. Greater reductions increase costs sharply.
Reneau and Taylor (1979)	Sediment	Lavon Reservoir, Texas (1,930 km²)	Enumeration	USLE delivery ratios, entrapment in flood control structures	Total costs (farm income losses plus government subsidies) of erosion control plans exceed benefits from abatement of offsite sediment damages.
Onishi and Swanson (1974)	1. Sediment 2. N in percolation	Forest Glen, Illinois (5 km²)	LP	USLE, delivery ratios for sediment, functional transport model for N	1. Assessment of farmers for offsite sediment damages reduces farm sediment losses 2. Sediment restrictions do not also control N losses. 3. Reduction of N in percolation to 10 mg/l severely reduces farm income.
Casler and Jacobs (1975)	P in runoff	Fall Creek, New York (330 km²)	LP	Functional transport model based on USLE	1. Without conservation practices, reduction of P losses greater than 10–20% substantially decreases farm income. 2. With conservation practices, increased feed purchases, P losses reduced 30–40% at small cost.

Wineman et al. (1979)	Water quality (subjective evaluation)	Black Creek, Indiana	None	Functional transport models	Nonpoint-source controls have non-uniform impacts on water quality; water quality/economic tradeoffs of policies differ among water quality problems.
Williams and Hann (1978)	1. Sediment 2. N, P in runoff	Little Elm Creek, Texas (101 km^2)	LP	Discrete simulation transport model	The study demonstrated that a decision theory framework for evaluating non-point-source pollution controls is in principle feasible.
Tseng (1979)	N in streamflow	Fall Creek, New York (330 km^2)	LP	Discrete simulation transport model	Watershed land use plans based on re-commended conservation practices reduced N export in streamflow.
Scherer (1977)	River salinity	Hypothetical 4 district example	DP	Discrete simulation transport model	Reallocation of water rights among water users by a market mechanism can increase total returns from irriga-tion water use.
Bardaie (1979)	River salinity	Imperial Valley, California	DP, LP	Discrete simulation model	Long-term management of salinity problems is possible with a dynamic program of changing diversions and crop selections from year to year.

TABLE 6 Application of farm management models.

Reference	Environmental emphasis	Location	Optimization procedure	Method of pollution estimation	Policy implication
Smith et al. (1979)	Sediment	New York, Iowa, Texas	LP	USLE, delivery ratios	1. Sediment losses from farms can often be economically controlled by concentrating control practices on field with high sediment delivery. 2. Erosion and sediment control plans are often substantially different.
White and Partenheimer (1979)	Erosion	Pennsylvania	LP	USLE	Conservation plans developed by US Soil Conservation Service usually decrease farm income (even with cost-sharing). These plans often require less profitable rotations.
Miller and Gill (1976)	Erosion	Indiana	LP	USLE	Uniform erosion controls will have differential effects on farm income. Small farms and farms with erosive soils will be most adversely affected.
McGrann and Meyer (1979)	1. Erosion 2. Pesticide use 3. Fertilizer applications	Iowa	LP	USLE	1. When erosion control requires rotation changes, costs can be substantial. 2. Income effects of erosion controls depend on soil resources. 3. Cost-sharing programs for structural measures are often inefficient. 4. Fertilizer reductions have more adverse impacts than pesticide bans.
Coote et al. (1975, 1976)	1. Erosion 2. Nutrients in runoff	New York	LP	Functional transport models	The effects of waste management regulations on a farmer are primarily determined by soil resources.
Haith and Atkinson (1977)	1. Erosion 2. Nutrients in runoff	New York	LP	Functional transport models	Greater farming intensity as indicated by number of cows/ha increases erosion and nutrient losses primarily due to more intensive cropping practices.

Service. These plans were compared with unrestricted plans for profit maximization and plans based on soil loss constraints. Both of the latter plans provided more flexibility to the farmer and in most cases generated more income than conservation plans. As was also seen by Smith et al. environmental controls exert their most adverse effects on income when they force changes in crop rotations.

Results from the modeling of four farms in Indiana (Miller and Gill, 1976) demonstrate the distributional effects of pollution control policies seen in the regional planning applications. Farm size and soil resources influence the impacts of control programs on the farmer. The work on McGrann and Meyer (1979) confirmed these observations, and indicated that government cost-sharing programs often do not encourage efficient (cost-effective) erosion control programs. A similar conclusion was reached by Smith et al.

The final two models in the table provide more complete descriptions of farm pollutant losses since they include estimates of nutrient losses in runoff. The estimates are determined by functional transport models based on the ULSE or US Soil Conservation Service runoff equation. The work of Coote et al. (1975, 1976) was designed to evaluate the effects of proposed manure management regulations on income and soil nutrient losses from dairy farms. It was found that the regulations did not necessarily reduce pollutant losses but could, depending on a farm's soil resources, decrease farm income. The model developed by Haith and Atkinson (1977) was a simpler version of the Coote et al. model and was used to investigate the effects of dairy farming intensity, measured in cows/ha on soil and nutrient losses. Although losses increased with intensity, the effect was caused more by cropping changes than the disposal of additional quantities of manure.

4 CONCLUSIONS

Two principal groups of mathematical models are available to aid in the analysis of agricultural nonpoint-source pollution. Chemical transport models estimate chemical losses from croplands to water bodies. Planning and management models are used to evaluate tradeoffs between environmental and agricultural production (economic) objectives. The development and application of these models have been rapid and haphazard. This paper is both a review of the current status of modeling activities and an attempt to establish a coherent framework, or classification based on model attributes, that can be used to compare and evaluate alternative modeling approaches.

Three types of chemical transport models are apparent. Continuous simulation models describe basic chemical transport processes with differential equations that are subsequently solved by analytical or numerical techniques or specialized computer languages. These models are generally applied to estimation of chemical losses in percolation or groundwater. Most continuous simulation models must be calibrated and are difficult to solve for realistic field conditions. At present, the modeling approach is perhaps best described as theoretical.

Discrete simulation models comprise the second, most common, type of chemical transport models, and describe transport processes with sequential algebraic equations based on water and chemical mass balances. Several of these models are operational tools for water quality planning since they are computationally efficient, do not require extensive data, and have been tested in field or watershed applications. However, none of the models has been tested extensively, and a great deal of further work is necessary before discrete simulation models of chemical transport can be routinely applied to agricultural pollution problems.

The final group of chemical transport models consists of functional models that do not attempt to simulate transport processes. Rather, the models are simple empirical or intuitive equations that predict chemical losses based on minimal data requirements. It is not surprising that these models have been widely used since they do not require extensive resource commitments. However, the accuracy of functional models is largely unknown, and they may not be reliable means of evaluating nonpoint-source controls.

Planning and management models are in principle the most useful models for policy-making since they provide estimates of economic and water pollution impacts of management practices. All such models are based on budgeting approaches which are usually solved by linear programming. Modeling applications are classified within three groups. Regional impact models are used for macroscale studies of farm and consumer income. Applications of such models in the USA have suggested that environmental controls on agriculture will have little impact on national or cornbelt income, but will increase prices to consumers and change regional crop distributions. Watershed planning models are applied to specific water quality problems and evaluate impacts of management practices, subsidies, and taxes on pollution and farm income. Farm management models evaluate the impacts of pollution control on the activities of individual farmers. Applications of these models have indicated that economic impacts will differ markedly from farm to farm.

Planning and management models have provided useful policy-making information. However, the economic components of the models are much better developed than components for prediction of pollution. The majority of models are based on the Universal Soil Loss Equation (USLE), and water quality impacts are limited to sedimentation estimates by empirical delivery and transport relationships. This is due largely to the limited availability of tested chemical transport models, and it can be anticipated that as these models become more reliable and generally accepted, more refined pollution estimates will be incorporated into planning and management models.

If the control of agricultural nonpoint-source pollution remains an important element of environmental planning, it is clear that much remains to be done in the development and testing of mathematical models. Models provide the quantitative information required for rational policy-making and in many, if not most situations, models will be the only feasible means of generating this information. In a relatively short time, scientists, engineers, and economists have produced a substantial body of work which will provide the basis for sustained future efforts. It is important for researchers, practitioners, and policy-makers to realize, however, that only a modest beginning has been made. The control of point-source discharges of wastewaters is based on more than a hundred years of research and testing, and a continued investment in nonpoint-source models will be necessary to establish a comparable level of technology.

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