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MODELLING AGRICULTURE RUNOFF:

OVERVIEW

by

J. Marsalek

Rivers Research Branch
National Water Research Institute
867 Lakeshore Road
Burlington, Ontario, L7R 4A6

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J. Marsalek
Rivers Research Branch, National Water Research Institute, Burlington, Ontario, Canada L7R 4A6

ABSTRACT Approaches to the modelling of agricultural runoff pollution are examined, starting with the modelling objectives and the level of analysis, followed by model implementation, and overview of existing models. Further refinements of the modelling process and of the existing models are proposed.

INTRODUCTION

In spite of recent improvements in point source control by technology-based effluent limitations, in many waters, the water quality goals and designated uses are unattainable without some control of nonpoint source pollution (Anon., 1986; Humenik et al., 1987; Vigon, 1985). Consequently, the abatement of nonpoint source pollution is attracting more attention in water quality management as one of the major control options. Among nonpoint sources, agriculture is generally recognized as the most widespread primary cause of water quality problems in both rivers and lakes (Anon., 1986). Environmental and economic damage, caused by agricultural nonpoint pollution in the U.S.A., was estimated at six billion dollars annually (Duda, 1985). Consequently, much attention in the analysis of nonpoint pollution has been directed toward agricultural activities and particularly agricultural runoff.

In the analysis of agricultural runoff pollution, the preference is given to computer models because of the scope and complexity of the underlying issues. From a broader perspective, such models should be viewed as one of the components in an integrated system of models used in river basin management and their main function is to provide loading inputs to the receiving water quality models (Ongley et al., 1988). Considering the high resource requirements for nonpoint modelling, there is a great need for rationalization of the modelling process. Toward this end, several reviews of agricultural runoff models were published recently and reflected the views of modellers (DeCoursey, 1985; Donigian & Rao, 1988; Leavesley et al., 1988; Novotny, 1986; Reckhow et al., 1985; Rose et al., 1988) or users (Bailey & Swank, 1983; Crowder, 1987; Oliver et al., 1988; Shaw & Falco, 1988). The paper that follows synthesizes and further expands such views in a broader context of water management.

FACETS OF AGRICULTURAL RUNOFF MODELLING PROCESS

System to be modelled

Proper analysis of agricultural nonpoint sources requires consideration of the complete agro-environmental system (Bailey & Swank, 1983). This system is rather complex and represents a combination of hydrological, ecological, agronomical, social, and economic subsystems. Pollution outputs from this system result from the summation of the responses of its subsystems and the watershed level response includes transfers from land to water, land to atmosphere, and atmosphere to land.

Need for modelling

The approaches to agricultural runoff pollution include a purely empirical approach, based on field observations and extrapolations and transpositions of field data to other watersheds, or the use of predictive models. The latter approach is generally preferred, because such models represent the most effective means for defining cause and effect relationships; extending the utility of limited field data; and, identifying, selecting, and implementing best control technologies (Bailey & Swank, 1983). At the same time, modelling should be recognized as part of an integrated approach to water quality management which includes planning, mediation, education, implementation, monitoring, and enforcement (Crowder, 1987).

Modelling objectives and level of analysis

In a top-down approach, the selected modelling objectives and attributes reflect the needs of water management planners and decision makers. Ideally, the main selection criteria should be the effects of modelling results on improved decision making and the costs of model application. Difficulties encountered in practice often lead to the replacement of such criteria by surrogate attributes of model appropriateness, resolution and uncertainty (Reckhow et al., 1987). According to the modelling objectives, three distinctive types of modelling are recognized; screening, planning, and design modelling (Barnwell & Krenkel, 1982).

Screening modelling is used for evaluation of broad policy measures, such as appraising agricultural pollution loads relative to other source loads, or the targeting of problem subareas. The corresponding models have to be simple, inexpensive to run and use inputs from the existing data bases. The predictions are done for long-term periods, assuming average or steady-state conditions, and large and diverse geographical areas (Bailey & Swank, 1983). The analysis may extend to stream transport, water quality in lakes or estuaries, and possibly the environmental fate of pollutants (Barnwell & Krenkel, 1982).

Planning modelling focuses on the development of basic features of pollution control strategies needed in subareas identified by screening modelling. Such subareas may be still fairly large, but relatively homogeneous and well characterized (Bailey & Swank, 1983). The pollution problems are defined in terms of types of pollutants and their impacts, and the choice of control options is reduced. Compared to screening modelling, more detailed, calibrated and verified models are needed to examine long-term water quality impacts (Barnwell & Krenkel, 1982).

Design modelling precedes the implementation of the remedial action plan. It is done for very specific conditions, using high spatial and temporal resolutions, and its ultimate goal is to devise a near-optimum solution to water quality problems.

Model selection

Before proceeding with model selection, the level of modelling needs to be established. It generally depends on the accuracy requirements of the analysis, characteristics of the system and contaminants studied, and available data and resources. For a selected modelling level, the required model attributes can be established and a suitable model is selected using the information given in the literature (Crowder, 1987; DeCoursey, 1985; Donigian & Rao, 1988; Novotny, 1986; Rose et al., 1988; U.S. EPA, 1987). The modelling process is often approached in a hierarchical way; screening modelling is conducted first and then followed up by more detailed planning or design modelling (DeCoursey, 1985). In the selection process, the costs of model application, in terms of input data collection and actual model running, need to be also assessed. Among models with comparable representation of important processes, preference is given to models with lower costs of application (Reckhow et al., 1987).

Model application

The modelling process consists of three phases comprising preparation of input data, model validation or testing, and the analysis of alternatives. The level of effort required for these tasks depends on the type of modelling and the model selected.

Preparation of model inputs includes assembly of input data and evaluation of model parameters. This task is simplified for screening models which use readily available data. For distributed models, the evaluation of physically based parameters may be relatively straightforward, but tedious, if large numbers of elements are used. In lumped models, some parameters have to be evaluated by calibration. Model validation consists of calibration and verification. Model calibration, which is particularly important for lumped models, is defined as a test of a model with known input and output information that is used to adjust or estimate factors for which data are not available (Donigian & Rao, 1988). Following calibration, the model should be verified against an independent data set, but this is often prevented by the lack of data. The calibrated and verified model can be used with good confidence to simulate alternative remedial measures, as long as the predictive conditions do not greatly exceed the range of conditions considered during calibration. The selection of the best remedial alternative is expedited by the use of decision-oriented models which provide cost information for remedial measures.

Implementation of results

Besides the technical issues discussed earlier, the implementation of the recommended remedial measures involves many socio-economic factors including planning; public information, education and acceptance; financing; mediation; monitoring and enforcement; and, post-audit. In the post-audit anaysis, the predicted and actual results are compared for the purpose of evaluation of the planning methodology including modelling. In this process, the proposed solutions are fully evaluated and, if necessary, altered to achieve project objectives. Post-audit findings may lead to discoveries of erroneous assumptions in the modelling and lead to changes in the project and management policies (Ongley et al., 1988).

OVERVIEW OF EXISTING MODELS

The selection of the agricultural runoff models, reviewed in this chapter, focused on fully operational, well-documented models which are used in Canada and the U.S.A.

Approaches to modelling

The basic processes considered in agricultural runoff modelling can be arranged into three groups dealing with hydrology, sediment, and agricultural chemicals. Such groups also cover agricultural practices and remedial measures. In decision-oriented models, it is further desirable to consider costs of remedial measures.

Hydrological models represent the basic component of agricultural runoff models, because the entrainment, transport, and fate of sediment and agricultural chemicals is controlled by the volume and rate of water movement through and across the soil surface (Leavesley et al., 1988). Possible contributions of subsurface runoff to surface runoff pollution loads are typically neglected (U.S. EPA, 1987). The basic processes considered in hydrological modelling include infiltration, runoff generation, and water balance.

Infiltration divides incoming precipitation between surface and subsurface flow components. Rates of infiltration depend on precipitation intensity, soil properties and water content, and surface effects. Both empirical and fundamentally based (i.e., processes based) approaches are used in infiltration modelling, depending on the availability of supporting data. At present, the use of the empirical Soil Conservation Service (SCS) procedure, in which infiltration equals precipitation minus runoff, is the most widespread method. Further development and refinement of fundamentally based approaches is expected to produce generally applicable methods. The main problems in modelling infiltration include spatial variability and cultural and natural surface effects, which include tillage, and surface crusting and freezing.

Three types of flow may contribute to generation of runoff, Hortonian overland flow, saturation overland flow, and subsurface flow. The mix of these flow components then affects the quantity, quality and timing of runoff. For daily precipitation data, the SCS approach is commonly used in runoff modelling. For more detailed data, the rainfall excess is determined and routed by physically based methods (Leavesley et al., 1988; Rose et al., 1988). The flow depth and velocity, and snowmelt may be important for sediment transport.

Water balance processes involve the storage and transmission of water in the soil profile and the concomitant movement of agricultural chemicals. The main mechanisms for water removal are evaporation from the upper layers and transpiration from the root zone. Models of soil water consider up to ten connected zones in the soil profile. Higher numbers of zones may improve the approximation of natural processes, but increase the application difficulties because of increased demands on field data.

Water-caused erosion involves detachment, transportation, and deposition of soil particles. Any of these processes may dominate under certain circumstances. Both empirical and fundamentally based approaches are used to model soil detachment. Among such approaches, the empirical Universal Soil Loss Equation (USLE), which was developed to estimate interrill soil losses over extended time periods, is the most common. With modifications, the USLE is also used for event erosion modelling (Leavesley et al., 1988) and, in conjunction with delivery ratios, for estimation of sediment yields.

Fundamentally based approaches to soil detachment distinguish between interrill erosion, caused by raindrops impact, and rill erosion caused by shear stress. The main advantage of such methods, the possible extrapolation of results, is somewhat offset by the need of calibration. In modelling sediment transport, difficulties are caused by applications of conventional streamflow equations to shallow surface runoff flows (Leavesley et al., 1988).

Hydrological and erosion processes control the transport of agricultural chemicals in surface runoff, and chemical processes control the types, forms and amounts of chemicals available for transport and their partitioning among water, sediment and air phases. Nutrient models need to address the magnitude, transport and redistribution of the water soluble, sorbed/desorbed, and sediment-attached nutrients at the storm time scale; and, in continuous modelling, the magnitude and transformation of the different nutrient fractions at the interstorm scale. The key chemical processes are those that control solution, sorption, and transformations between fractions. Approaches to nutrient modelling vary greatly. Even though defensible formulations of such processes have been proposed, the lack of useful and reasonable methods for estimating nutrient parameters causes serious difficulties (Leavesley et al., 1988). Consequently, simplifications have to be made by neglecting certain fractions and using such concepts as potency factors or enrichment ratios reflecting particle size selectivity in the erosion process (Novotny, 1986).

The transport of pesticides is a very complex process which at present is not yet amenable to a rigorous mathematical description. The chemistry of pesticides and their hydrological and sediment transport are very diverse. Main problems include pesticide dissipation at the soil surface, pesticide entrainment/extraction from the soil surface into runoff, and pesticide partitioning between soil and water. Although progress is being made in formulation of such processes, no existing pesticide models have proven accuracy and their best use is for comparisons of different scenarios.

Features of the representative existing models

A list of representative agricultural runoff models and their features is presented in Table 1.

Table 1. Representative Agricultural Runoff Models[1]

Model	Level of Analysis[2]			Hydrology			Water Quality				Time Scale			Spatial Scale			
ACTMO		X		X		X	X	X	X		X				X		
AGNPS	X			X			X	X		X	X			X	X	X	X
ANSWERS		X	X	X	X		X	X			X					X	
ARM		X		X	X	X	X	X	X	X			X			X	
CREAMS			X	X			X	X	X				X	X			
GAMES/P	X						X	X				X				X	
GWLF	X			X	X	X	X	X				X	X				X
HSPF		X		X	X	X	X	X	X	X			X			X	
LANDRUN		X		X	X	X	X	X					X			X	
LANDS	X						X					X					X
NPS		X		X	X	X	X	X		X			X			X	
SEDIMOT II		X		X	X		X				X					X	
SWAM		X	X	X	X		X	X	X				X			X	
UNIT A. LOADS	X						X	X	X	X			X				X
UTM-TOX			X	X	X	X			X	X			X			X	

References:
[1]GWLF - Haith & Shoemaker (1987), SEDIMOT II - Wilson et al. (1988).
[2]Some models could be used for both planning and design.

The information in Table 1 indicates that there is a wide range of agricultural runoff models available. Relatively simple screening models do not simulate hydrological processes, but produce sediment yields and/or chemical loads for events, or extended periods (a month, season or year). Examples of such models are AGNPS, GAMES/P, GWLF, LANDS and UNIT AREA LOADS. All these models provide results for large areas, require minimal input data, and generally do not require calibration.

The remaining models in Table 1 are used in planning and design. Typically, such models simulate basin hydrology, sediment yields and some agricultural chemicals. Most of these models operate in a continuous mode and are applicable to relatively small areas.

MODEL IMPROVEMENTS

Various interest groups, involved in nonpoint pollution modelling, perceive different needs for improvements in modelling. While the model builders emphasize the need for improved comprehensiveness and mathematical rigor of models (Leavesley et al., 1988), the users are more interested in better guidance for model selection and application, and extension of the capabilities of the existing models (Oliver et al., 1988; Shaw & Falco, 1988).

The widespread availability of models increases the possibility of their misapplications as a result of the lack of knowledge of model limitations (Oliver et al., 1988). Such occurrences could be reduced by improved model documentation which would list not only model capabilities, but also its limitations, minimum data requirements, robustness and range of applicability of key computational procedures, and the types of situations to which the model should not be applied.

Model applications and preparation of input data would benefit from wider use of geographical information systems and expert systems (Ongley et al., 1988; Shaw & Falco, 1988). Other improvements desired by users include the development or refinement of models to fully reflect end-user requirements, more user friendly software, better computer graphics, availability of methods for evaluation of modelling results, and better training.

The capabilities of the existing models should be further developed or refined to address such problems as planning level modelling in intermediate size basins (say up to 1,000 km), contributions of subsurface flow to runoff pollution, simulation of the effects of untreated or buffer zones in pesticide simulations, toxic chemicals leaching and downstream fate, proper modelling of environmental impact and risk assessment, cost/benefit analysis, and statistical analyses and interpretation of results (Oliver et al., 1988; Shaw & Falco, 1988; U.S. EPA, 1987).

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