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Modelling contaminant transport at catchment or regional scale

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Abstract

Most models for contaminant transport have been developed and tested at the scale of the field plot, the lysimeter or even the laboratory soil column. Modelling at the scale of the catchment or region therefore brings some interesting new challenges. The interaction between non-linearity in a model and variance in one of its parameters can cause errors in modelling even small areas of land; the reasons are explained and a test for non-linearity given. It is necessary to know whether this problem will intensify as the area modelled increases, and it is suggested that the variogram provides a means of answering this question. The intrinsic variability of the soil is only one of the factors that have to be considered at catchment or regional scale. At these scales, land use is likely to be the dominant source of variability in the loss of contaminants such as nitrate and phosphate. Simulating land use effects is essentially a data-handling exercise that has to involve the use of geographic information system with models for N turnover and leaching in what is essentially a decision–support system. Questions surrounding the development and use of such systems are discussed. Another extra dimension is introduced at catchment scale and above by the need to consider the subsequent fate of water after it has passed through the soil profile. It is necessary to model its partitioning between that which percolates into ground water and that which moves as surface runoff into streams and other surface waters, and possible approaches are described. The largest land use change in the UK this century has been the ploughing-up of old grassland. A brief account is given of the way the consequences were simulated using a simple model. The results suggest that the ploughing made a very substantial contribution to the nitrate problem in the UK. There is no fully satisfactory way of validating a model used at catchment or regional scale. All that can probably be achieved is to validate parts of the model for much smaller areas and to subject the whole model to a test of its general efficacy. © 1998 Elsevier Science B.V.

Keywords: Non-linearity; Variance; Variogram; Land use change

1. Introduction

Some models for solute transport in the soil were developed and tested initially against laboratory columns of sieved soil, others against undisturbed cores, lysimeters or field plots. There is, however, an increasing need for models or systems of models that describe the transport of environmental contaminants at catchment or regional scale, and there is an obvious need to consider carefully how models developed for small areas of land can be used for much larger areas, and indeed, whether it is wise to use them in this way. One issue that must be addressed is that of the intrinsic variability of soil properties used as parameters for models. This can be a problem at the scale of a single field; does the problem increase in proportion to the area under consideration? In addition to this natural variability, there is the extrinsic variability that results from land management and

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which is therefore, central to the topic of the conference.

A further complication that arises from the change of scale is the consideration that needs to be given to the fate of water passing through the soil. With lysimeters or field plots, it is normally sufficient to simulate the loss of water together with the concentration and thence the overall loss of the contaminant. At catchment or regional scale, however, requires integration of the losses of water and contaminant over the whole area, and this involves partitioning the flow of water between that which percolates through to the ground water and that which moves as surface runoff to streams and other surface waters. Concern may also be expressed with the route followed by the water, particularly in the latter case.

The term 'soil contaminants' could imply either substances that contaminate the soil or substances coming from the soil that contaminate other parts of the environment, particularly natural waters. Contaminants of the former kind are a problem because they are *not* readily transported from the soil. Thus, modelling contaminant transport concerns the latter category of contaminant, and the commonest causes of concern are nitrate, phosphate and pesticides. Any long-term changes are likely to involve the soil organic matter, because it is not only the source of a substantial proportion of the nitrate problem but also the most important sorbant for pesticides in the soil. Thus, it needs to be considered in conjunction with models for the turnover of carbon and nitrogen through organic matter. The sorption of pesticides is not discussed here, but the topic has been reviewed by Nicholls (1991).

This paper discusses firstly the intrinsic variability of the soil and explains why this is a problem particularly for non-linear models. Whether the problem intensifies as the scale shifts from small to large areas is an important question and is also discussed, with particular reference to leaching models. The flows of water and solutes from these models have to be partitioned between percolation into groundwater and runoff into streams and other surface waters and models for this process are described. The effects of land use pose a problem that is rather different from those posed by other forms of variability, and modelling them is suggested to be in part, an exercise in data management. The other part of this problem lies in modelling, at an appropriate level of complexity, the turnover of carbon and nitrogen through the soil organic matter and the behaviour of phosphate. Finally, the long-term modelling of the largest change in UK land use this century, the ploughing-up of old grassland during and after World War II, is discussed.

2. Non-linearity, variance and scale

2.1. The interaction between non-linearity in models and the variances of their parameters

The problem of non-linearity is best approached through some very simple examples. Table 1 shows some functions of x evaluated for replicated values of x in two ways, either by evaluating the function of each x and then taking the mean, or by taking the mean of x and then evaluating the function. For only one of the four functions do the two procedures give the same result. Plotting the functions on a graph would show that only the first one, $f(x) = 3x$, is linear. The others, which show the discrepancy between the procedures are non-linear. Differentiating these functions would show that the first one is also the only one to have a zero second differential with respect to x. The relevance of this point is shown by some equations of Rao et al. (1977) which relate the mean of a function $f(x, y)$ of x and y to the means

Table 1
Evaluation of some simple functions of x for replicated values of x. Two procedures: evaluate $f(x)$ for all x then take mean, or take mean of x then evaluate $f(x)$

Function	Procedure	Result
$f(x) = 3x$	evaluate $f(x)$ first	6.0
	take mean first	6.0
$f(x) = x^3$	evaluate $f(x)$ first	12.00
	take mean first	8.00
$f(x) = 5/x$	evaluate $f(x)$ first	3.05
	take mean first	2.50
$f(x) = e^x$	evaluate $f(x)$ first	10.06
	take mean first	7.39

Values of x 1, 2, 3. Mean = 2.

of x and y, μ_x and μ_y, and their variances σ and σ. The mean of the function is

$$\mu_{f(x,y)} = f(\mu_x, \mu_y) + C. \quad (1)$$

This shows that, as in Table 1, the result of evaluating the function before taking the mean is not the same as taking the means of x and y before evaluating the function. The term C, which is the difference between the two procedures is given by

$$C = \frac{1}{2}\left\{\left(\frac{\partial^2 f(x,y)}{\partial x^2}\sigma_x^2\right) + \left(\frac{\partial^2 f(x,y)}{\partial y^2}\sigma_y^2\right) + \rho\left(\frac{\partial^2 f(x,y)}{\partial x \partial y}\right)\sigma_x\sigma_y\right\} \quad (2)$$

where ρ is the product moment correlation between x and y. It can be seen at once that the difference between $\mu_{f(x,y)}$ and $f(\mu_x, \mu_y)$, that is, the difference between the two procedures for evaluation, arises from the second partial differentials of the function with respect to x and y, and that this difference occurs only if x and y have non-zero variances.

The simple non-linear functions shown in Table 1 can all be transformed to linear functions, but this is not an option for all but the simplest computer models. It is certainly not an option for transport models for the unsaturated zone that are based on the Richards Equation and the Convection–Dispersion Equation and which are therefore non-linear. If a model is non-linear and one of its parameters varies in space such that it is properly represented by a probability distribution, there are certain consequences: (1) The mean of an output from the model will depend not only on the mean of the parameter but also on its variance. This emerges from Eqs. (1) and (2) and it carries the implication that there is a risk of an inaccurate result when a non-linear model is used with a single-valued parameter based on a soil property known to be spatially variable. (2) With kriging and other linear spatial interpolation procedures the results of interpolating the parameter before running the model will not be the same as those of running the model first and interpolating the output (Addiscott and Bailey, 1990; Stein et al., 1992). This may raise uncertainties when non-linear models are used for simulations over substantial areas. If the way in which the model is used carries the implicit assumption that using spatially interpolated or averaged parameter values is equivalent to applying the interpolation or averaging procedure to the model output, the results could be misleading. (3) Procedures for validation and parameterisation could be compromised. This problem is discussed elsewhere with some simple examples by Addiscott et al. (1995).

It is clearly important to know whether a model is non-linear with respect to one or more of its parameters. If the model is in the form of a differentiable function, it can be tested simply by checking whether its second partial differentials with respect to its parameters are zero. If they are not zero, the model is non-linear. An alternative test is needed for the majority of models that are not in a form in which they can be differentiated. Eq. (2) shows that a non-linear model, with non-zero partial differentials, will be sensitive to changes in the variance of its parameters. The model can therefore be tested for non-linearity by increasing the variance of each of its parameters (Addiscott and Tuck, 1995). If the mean of an output from the model changes when the variance (but not the mean) of the parameter is increased, the model is non-linear with respect to that parameter. The degree of non-linearity, shown by this test may vary according to which output from the model is tested (Addiscott and Tuck, in preparation). It will often be useful to perform this test for two or more parameters and compare the results, and because the parameters may differ in their orders of magnitude, it is helpful to plot the mean of the parameter against the coefficient of variation of the parameter rather than the variance (Fig. 1). This test, obviously needs the models to be run with their

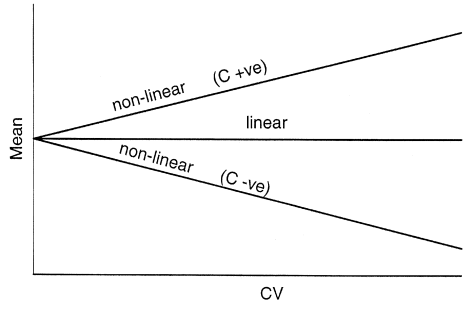

Fig. 1. Testing for non-linearity. Plotting the mean of the output against the CV of the parameter (from Addiscott and Tuck, 1995).

parameters as probability distributions. This can be achieved by using either a Monte-Carlo routine or the 'Sectioning Method' of Addiscott and Wagenet (1985).

2.2. Changes in variance with scale

It is necessary to ask how the problems that arise from the interaction between the non-linearity of a model and the variances of its parameters will affect the use of models at catchment or regional scale rather than that of the plot or field. In effect, this is asking how parameter variance changes with increasing distance. This can be answered technically by knowing the covariance function, or alternatively, the variogram of the parameter. There will be an underlying variogram of the random process, representing the inherent variability of the soil property used as a parameter, for which there will be a true realization within a defined (geostatistical) region. This is estimated through the experimental variogram, which describes the variation in data, and this in turn has to be quantified by a model fitted to the data. The result shows through diagrams, such as those in Fig. 2, the way in which the variance of the parameter increases with the lag distance between the points at which it is measured.

The key distinction in the present context is between *bounded* and *unbounded* variograms. 'Bounded' implies that the semi-variance increases with the lag distance up a maximum value known as the *sill* beyond which there is no further increase. The lag at which the sill is reached is known as the *range*. Of the geostatistical models used to describe variograms the simplest is the bounded linear model (Fig. 2), in which the semi-variance increases linearly with the lag distance until the sill is reached. For soil properties the semi-variances often approaches the sill curvilinearly, making other models, described by Webster and Oliver (1990), more appropriate. An unbounded variogram has no sill and therefore no range, and will often be described by a power function, of which the straight line is the simplest.

Many experimental variograms appear to have appreciable semi-variance at zero lag. This is known as '*nugget*' variance. Indeed, some variograms appear as pure nugget variation, probably because the shortest sampling interval is longer than the range of the variogram.

It may not be obvious how the variogram could influence the mean of an output from the model, but attempts to show the implications of the variogram are shown in Figs. 2 and 3. Fig. 2 gives examples of variograms, and Fig. 3 gives a simplistic illustration of their effects using the simple function $f(x) = e^x$. This is not an exact illustration, because x is not

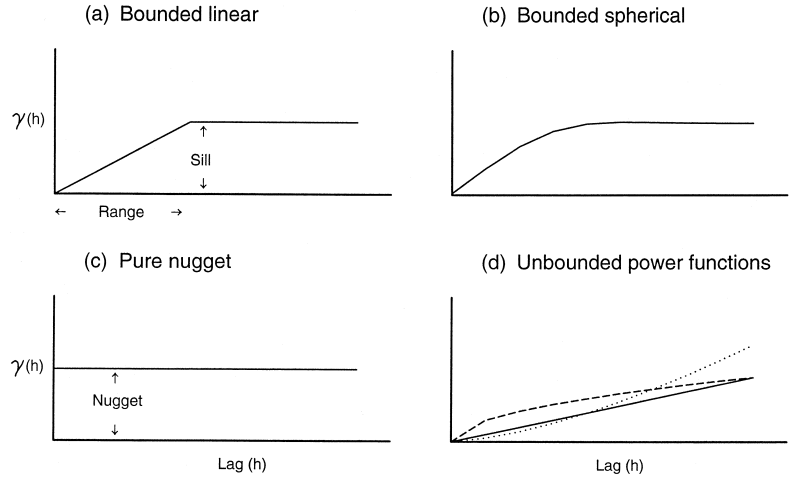

Fig. 2. Examples of variograms showing sill, range and nugget variance (from Addiscott and Tuck, 1995).

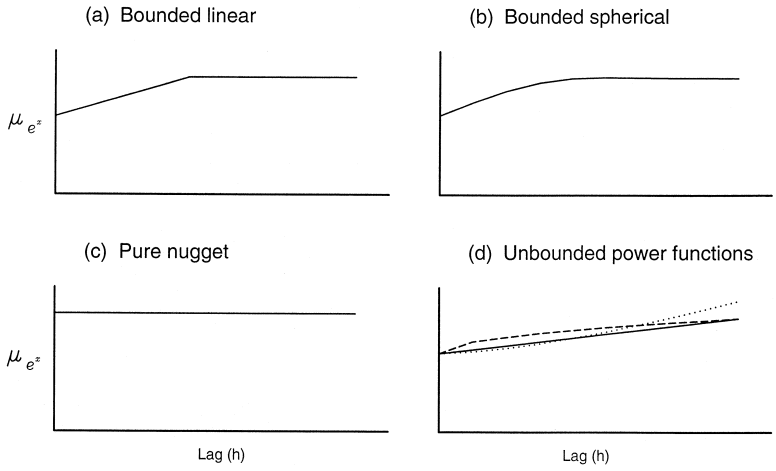

Fig. 3. Consequences of the variograms shown in Fig. 2 for the mean of the simple function e^x. For simplicity it was assumed that $\mu_x = 1$ and the sill of the variogram was 1 (from Addiscott and Tuck, 1995).

strictly a parameter. However, the function is convenient for the purpose because its second differential is also e^x; Eq. (2) shows that the second differential is at the root of the influence that the variance, and thence the variogram, exerts on the mean of the output. In Fig. 3, the mean of the function shows a component arising from the mean of x, which is fixed, and a component resulting from the variance of x, which changes with the lag for all except the pure nugget variogram. (Fig. 3 was prepared with the very simple assumptions shown in the Legend, so no units are shown.) The key point here is that in principle, the variogram of a parameter influences the mean of an output from a non-linear model, and so, needs to be considered when the scale at which such a model is used changes.

Fig. 3 shows clearly that whether the variogram is bounded or unbounded is important. With a bounded variogram, the variance and its interaction with the non-linearity of the model all increase over the range of the variogram, that is, until the sill is reached. Consequently the mean of an output from the model will change too. If the range of a parameter can be contained within a field, there is no reason to expect problems when the model is used over areas greater than a field. If, however, the parameter has an unbounded variogram, both the variance and the problems it causes, could in principle increase indefinitely as the model was used over increasingly large areas.

It may be necessary to consider, not only whether the range of a parameter can be contained within a field, but also whether there is a change of soil type at the field boundary, as this would obviously be relevant. Some field boundaries exist because experience showed the farmer over a period of time that two parcels of land had differing soil and needed to be managed in different ways. Others are there because of landscape features such as changes of slope and water courses. The majority, however, are there because the land was divided for purely human considerations, such as inheritance, enclosure or land allocation. It can often be seen quite clearly from the air that the neat rectangular field divisions bear little relationship to the variability in the soil.

3. Modelling at catchment or regional scale

3.1. Models for simulating the physical process of leaching at catchment or regional scale

Two previous papers (Addiscott, 1993, 1994) suggested that, for use over large areas, certain characteristics were desirable in a solute transport model.

1. The model should be linear with respect to its parameters.
2. The parameters should not be too variable.
3. Parameters that are spatially correlated should be additive.

These papers concluded that models with capacity parameters, such as those of Burns (1974), Addiscott (1977) and Corwin et al. (1991), should be useful for simulations over large areas because there should be few problems with the interaction between non-linearity and variance. These models have not been tested specifically for non-linearity, but their general nature suggests that they are likely to behave linearly. Also, the capacity of the soil to hold water varies little in space and commonly has a coefficient of variation of about 10 percent. It is also additive in nature. Because non-linearity is not really a problem, the nature of the variogram for capacity parameters is not really an issue, but it is worth noting two reported studies. Warrick et al. (1990) measured the variogram of soil moisture content on two occasions at the same site, obtaining bounded variograms described by a spherical geostatistical model. They computed the range of the variogram with various sampling arrangements, obtaining values for the range from 245 m to 273 m. Stein et al. (1989) measured the variogram for available water in the soil profile and fitted an exponential geostatistical model with an effective range of 280 m. Whether ranges of these orders can be contained within a field obviously depends on its size but the variance of capacity parameters seems unlikely to increase much beyond field scale.

More mechanistic models that invoke the Richards Equation and the Convection–Dispersion Equation are well known to be affected by the interaction between non-linearity and parameter variance, but the information available suggests that these problems are not likely to be compounded when the variogram is introduced as a consideration. Banton (1993) found pure nugget variograms for the saturated hydraulic conductivity, K_s, in measurements both in situ and on cores removed to the laboratory, probably because the area within which the variation in K_s was contained was very small. Beven and Germann (1982) showed that the length of a representative elementary volume for microporosity is about 1 cm. It is rather larger, 1–10 m, for macroporosity. The unsaturated hydraulic conductivity, $K(\theta)$, depends on the volumetric moisture content, θ, as well as K_s, and is not usually measured directly. Given that K_s shows nugget variance and θ a bounded variogram, it seems unlikely that the variance of $K(\theta)$ will increase beyond field scale. Other questions, notably those raised by Beven (1989), seem likely to be more important in assessing the viability of physically-based mechanistic transport models over large areas.

Petach et al. (1991) used the LEACHM mechanistic solute leaching model (Wagenet and Hutson, 1989) in conjunction with a geographic information system (GIS) to simulate pesticide movement through the soil of a large area of land and suggested in their conclusion that a simpler model should be considered for the purpose. This suggestion was taken up by Hutson (1993), who used a capacity-type model in a similar context. The capacity-type TETrans model of Corwin et al. (1991) has been used in a GIS context, by Vaughan and Corwin (1994), to simulate the movement of water and salts in 2350 ha of the San Joaquin Valley of California.

3.2. Modelling the fate of water and contaminants when they leave the soil

Traditionally, modelling flows of water within the soil profile has been the province of the soil physicist, and modelling flows of water in the catchment has been the responsibility of the hydrologist. The connection between the two flows has not always been made very effectively. Part of the problem may lie in philosophical differences between the two disciplines. Physicists generally take a rigorous, mechanistic approach and some are suspicious of modelling (e.g., Philip, 1991). Some hydrologists also tend to take a physical approach, but others have reacted to the complexity of the problem by taking a 'systems' or 'black box' approach, which relies on the establishment of input–output relationships.

Broadly speaking, catchment hydrology can be divided into three parts:

1. Surface hydrology,
2. Soil profile hydrology,
3. Groundwater hydrology.

The surface hydrology has a key influence on the whole because it determines the eventual fate of the water. This again is divided into three parts:

1. Surface run-off,
2. Subsurface run-off,
3. Baseflow.

The surface run-off needs to be divided between overland flow and channel flow. The subsurface flow is the proportion of the rainfall that infiltrates the soil and then moves laterally through the soil until it reaches a stream channel.

In the system currently under development at Rothamsted, the area of catchment is divided into cells and each cell is marked as active or inactive. Obviously, only active cells contribute to catchment flows. Streams can be defined by marking the cells through which they flow. The direction of flow within a given cell is determined by the topology of the landscape.

Overland flows can either be simulated or ignored. If they are to be simulated, the method of Mein and Larson (1973) or Hillel (1977) is used to compute the infiltration capacity of the surface layers and any excess rain moves overland to the stream. Flow of water and solute within the soil profile is simulated by a simple leaching model derived from the SL model (Addiscott, 1977) and SLIM (Addiscott and Whitmore, 1991). Output from the profile leaching model is fed into the three-dimensional water and solute model, HST3D, of the United States Geological Survey. Stream flow and the solute concentration within it are simulated using WASP or QUAL2E.

This has necessarily been a very brief account of the problems of simulating the flows of water and solute. Further information on the problem of modelling water flows at the catchment scale has been given by Beven (1989, 1991).

3.3. Modelling the effects of land use

So far, discussion has been confined to the intrinsic variability of soil properties and the problems to which this variability gives rise. When large areas of land are considered the extrinsic effects that arise from land use become more important. This becomes obvious when moving from modelling the processes within one field to modelling them for a whole farm. Not only will different parameters be needed for the crops in each field, or even different models, perhaps, if energy crops are grown, but consideration may need to be given to transport of nutrients and other materials between different parts of the farm. Modellers become involved not only in modelling but in handling information. At the catchment scale, may be encountered, farms with different land use policies and also woodland or heathland that is left to 'nature'. With large catchments or regions, there may be substantial areas of urban land, for which it is important to know how domestic effluents are dispersed, given that each individual excretes 6 kg N per year (The Royal Society, 1983).

Moving up from the field scale, also needs knowledge of how much area is assigned to a given crop or land use and where the use occurs in relation to the flows of water within the landscape. Here again it is no longer a straightforward modelling problem but a problem of handling information. This is a problem for which help is at hand in the form of the geographic information system (GIS).

Two examples were given above of the use of leaching models in conjunction with GIS, but neither of them dealt with the leaching of nitrate or that of phosphate. To model nitrate losses at catchment or regional scale requires a *system* that comprises models for the physical leaching process and the biological processes that mediate the turnover of carbon and nitrogen through the soil organic matter, together with data handling facilities that provide parameters for different crops and soil types and information about farming operations. The system also needs to incorporate a GIS for the reasons outlined above. What is being described is, of course, a decision support system (DSS).

There is considerable interest in the development of such systems and at Rothamsted scientists are currently working on one that is intended to combine the SUNDIAL nitrogen turnover model (Bradbury et al., 1993) with one of the simple leaching models developed at Rothamsted. One interesting question that has arisen is the relation between the GIS and the rest of the DSS. Should the GIS control the whole system, should it interact with the models through the DSS, or should both the GIS and the models be 'embedded' in, and therefore, subservient to, the DSS? This problem has been discussed elsewhere and Fedra (1994) suggested that the DSS should be in control of an 'embedded' GIS, as in the HYDRA irrigation control system (Jacucci et al., 1993).

Clearly, a system as complex as a DSS need not be restricted to any one set of models, and if a

modular system is used, the DSS can provide models of different levels of complexity for use at different scales. Decision support systems are also needed for phosphate, particularly that from animal wastes, and one possibility for providing such a system lies in the ANIMO model (Schoumans, 1995).

3.4. Modelling the effects of ploughing-up old grass

The largest change in land use that has occurred in the UK this century, and probably for a very long time, has been the ploughing-up of old grassland that took place during and after the Second World War. The effects of this ploughing on the development of the 'Nitrate Problem' have probably been underestimated, and it is likely that they still need to be taken into account in assessing the origin of nitrate pollution in parts of the UK today. The most effective effort to quantify by modelling the influence of this ploughing on the concentrations of nitrate in water leaving the soil has been that of Whitmore et al. (1992), and a brief account of their study is given below.

Whitmore et al. extended their study to the whole of England and Wales, and therefore had to keep their model simple. They first fitted a simple exponential model to data from experiments in which the decline in organic N following the ploughing of grass had been measured. This took the form:

$$N_{org} = a + be^{-kt} \quad (3)$$

where N_{org} was the quantity of organic N (kg ha^{-1}) in the top 25 cm of soil at time t and a, b and k were constants. The information obtained from this curve was combined with information on the areas of old grassland that were ploughed each part of the country in each year. The largest decline in permanent grassland took place between 1940 and 1945 but the decline has continued intermittently in subsequent years. The model kept a running tally of the amounts of nitrogen coming out of organic matter in the years following ploughing, using the following equation:

$$N_j = \Sigma\left(A_1 L_j + A_2 L_{j-1} + \ldots + A_j L_1\right) \quad (4)$$

where N_j is the quantity of nitrogen (kg ha^{-1}) mineralized just from ploughed grassland in the jth year after ploughing, A is the area (ha) of grass brought under the plough each year and L is the loss of nitrogen each year (kg ha^{-1} yr^{-1}) obtained from Eq. (3).

As j becomes large (more than 20 yr), N_j approaches the value of a in Eq. (3), which was 3954

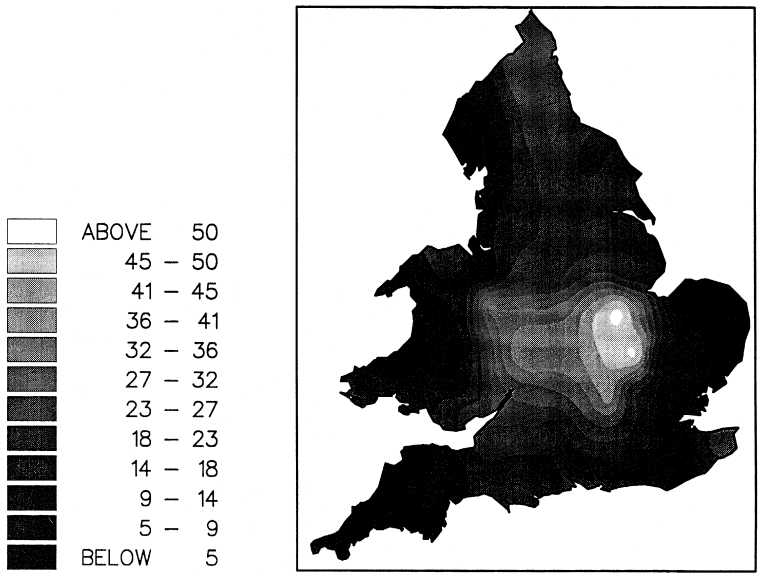

Fig. 4. Potential increase in the concentration of nitrate–N in water draining from the soil (mg N l^{-1}); grassland ploughed between 1939 and 1945 (from Whitmore et al., 1992).

kg for every ha of grass ploughed for all j years. This mineralization of almost 4 t N ha^{-1} given by the model agrees almost exactly with the mineralization estimated from losses of organic N where very old permanent grass was ploughed at Rothamsted.

The model assumed that the losses of N on ploughing would be in addition to the losses normally characteristic of arable agriculture, and that the N mineralized was nitrified and mixed with the water percolating through the soil. This annual loss, as given by Eq. (3), was divided by the mean quantity of effective rainfall (that is, rainfall less evaporation) on a county by county basis to give the potential contribution of the ploughing to the nitrate concentration in water draining from the soil each year. The results were expressed as maps such as that in Fig. 4. One of the more startling conclusions was that, in 1945, water draining from arable land anywhere in England and Wales could have exceeded the current limit for nitrate concentration set by the EC.

3.5. Validating models at catchment or regional scale

No model can be 'validated' in the sense that it has been unequivocally justified. All that can be shown is the probability that it has been refuted, which may be small but will always be finite. There is inevitably an element of subjectivity, because someone has to decide what level of probability is acceptable. Various statistical tests, such as those suggested by Whitmore (1991), can be used to assess the probability of refutal, and they necessarily depend on the replication of the measurements against which the model is validated or sufficient simulation-measurement pairs to permit the determination of the correlation coefficient and the mean difference.

Every catchment is unique, which makes it very difficult to assess the performance of the model statistically. One possibility is to measure contaminant concentrations at the outflow or at fixed points within the catchment and test whether the model reproduces these concentrations satisfactorily. This process, however, is also fraught with difficulty if the concentrations change rapidly. If the model gives the general pattern right but is out of phase with respect to the fluctuations of the concentration, results will be poor. Alternatively, if the tests are done at the outflow, errors in various parts of the catchment may compensate each other to give a spuriously acceptable result.

The best option may be a dual one. Validate individual parts of the model at a scale at which replication and statistical tests are feasible. Then subject the whole model to a test of efficacy, which means (literally) testing it to see if it gives the intended effect. A test of efficacy might include the introduction of a perturbation that occurs in the natural system. If the model responds to the way that the natural system does, this is a sign of efficacy.

4. Discussion

How can contaminant transport best be modelled at catchment or regional scale? The conclusion reached within the Soil Science Modelling Group at Rothamsted is that no single model will be appropriate, or even usable, for all purposes at all scales. There is a general inverse relationship between the size of the area to be modelled and the level of fundamentality and complexity that is appropriate in the model. Put another way, the model needs to be *functional with respect to scale* (Addiscott, 1993). The main reason for the relationship is that the most mechanistic models are usually the most likely to be non-linear and tend to have the largest demand for input information. For very large areas, such as the whole of England and Wales, quite simple models can be very useful, as was shown by Whitmore et al. (1992). At the other end of the scale, a fully mechanistic transport model may be needed to elucidate particular details or anomalies in the behaviour of solutes at particular points. Relatively simple capacity-type solute transport models seem most likely to be useful for simulations at catchment scale and they need to be allied to models of comparable complexity for biological N processes. With models of this kind, problems are not likely to arise from the interaction between the non-linearity of the model and the variance of the parameter. This permits concentration on what is likely to be the dominant form of variability, that arising from decisions taken about land use. This variability has to be treated in a way that permits the handling of information and data and this implies the use of a decision support system.

Such a system will ideally incorporate a choice of models for solute transport and N turnover processes that permits the use of appropriate models at particular scales.

For some purposes, it will be enough to assess the amounts of contaminant and water emerging from the soil within a landscape and derive from them an average contaminant concentration for the area, as was done for example by Whitmore et al. (1992), who assessed the concentration on a county basis. For other purposes, however, it is necessary to know the routes followed by the water carrying the contaminant before the environmental impact of the latter can really be assessed. This necessitates the bringing together of hydrological and soil science expertise. At present, there is no shortage of soil scientists' models for solute transport in the soil profile or hydrological models for catchment flows. What is needed now is for these two categories of models to be integrated. This should be achievable, but one problem that will have to be solved is that of the disparity between the time-steps used in the modelling of various processes. The SUNDIAL N turnover model (Bradbury et al., 1993) uses a time-step of a week and the SLIM leaching model (Addiscott and Whitmore, 1991) has a daily time-step. For many catchment models, however, an hour is as long a time as a week is in politics.

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