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Integrated management of sensitive catchment systems

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Abstract

Until recently, ‘land use’ was regarded as a single function: in rural areas of the UK this simply meant ‘farming’ or, in the uplands, ‘forestry’. However, there is now growing recognition of the multiple use of land, and farming or forestry must compete with other functions, in particular water supply. Links between hydrological pathways and stream water quality are described as a context for understanding the transport of pollutants to the river system. The concept of landscape sensitivity is then described and applied to the topics of soil erosion and nitrate leaching. Based on these analyses, guidelines for integrated management of sensitive catchment systems are proposed. © 2001 Elsevier Science B.V. All rights reserved.

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1. Introduction

The pathways taken by water draining to a stream determine many of the characteristics of the landscape, the various ways in which land can be used and the management practices required to sustain these uses (Dunne, 1978). At the same time, land use can significantly influence the hydrology of a river basin: catchments are susceptible to changes in land use, affecting both the quantity and quality of runoff produced (Newson, 1997). This paper focuses on water quality in rural catchments, although the approach adopted is equally applicable to studying the impact of land use change on water yield or the impact of urbanisation on catchment hydrology.

Until recently, ‘land use’ was regarded as a single function: in rural areas of the UK this simply meant ‘farming’ or, in the uplands, ‘forestry’. However, there is now growing recognition of the multiple use of land; farming and forestry must compete with other functions (Newson, 1997; Burt, 1999). In that practically all rural land lies within a catchment area, conflicts have almost inevitably arisen as modern agriculture has

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affected water supplies, especially in relation to pollution. Farming has become steadily more intensive as farmers have met the challenge of government policy, especially in relation to the provision of subsidies, and social trends. This has produced an increasingly capital-led industry, driven by the need for profit, and as a result, farmers have found it more difficult to sustain their position as stewards of the countryside. At the same time, there has been increased interest in the rural environment from many sections of the general public: some demand better access, others have an interest in nature conservation, while for many more, farmland simply provides a pleasing backdrop for a drive through the countryside. An integrated approach to land use planning in rural drainage basins extends beyond the narrow conflict between farming and water quality. Growing recognition of the multiple uses of rural land reflects the wide range of interests now operating in the countryside. The move from ‘cure’ towards ‘prevention’ as a more sustainable solution to water quality management requires involvement of all stakeholders with a legitimate interest in the outcome of the decisions being made. Farmers (and foresters) are now encouraged to manage their land in a way that limits pollution at source and delivers a range of environmental benefits (Burt, 1999).

The impact of modern farming is seen in two ways: on and off the farm. On the farm, concerns relate to soil degradation and to the loss of habitat for wildlife. Off the farm, impacts include pollution of potable water supplies, damage to aquatic ecosystems and flooding of property by soil-laden runoff. Nitrate leached from farmland is the main source of nitrate in surface and ground water (Burt et al., 1993). It is overly simplistic to censure the increased use of fertilisers, although this is partly to blame. Modern farming practices are also implicated, including larger numbers of livestock, new crops, and increased frequency of ploughing. Under-drainage of near-stream land has also been important in allowing arable crops to be grown on land that was previously waterlogged (Green, 1979). This has been doubly deleterious: large leaching losses from the riparian zone *and* the loss of buffer zone functions. Nitrate levels have risen steadily in many water supplies since the 1960s (Johnes and Burt, 1993), and in some cases are now perilously close to legal limits (50 mg l^{-1} NO_3 or 11.3 mg l^{-1} NO_3–N). The EU’s Nitrate Directive (91/676) requires nitrate levels to be maintained below the legal limit in *source* waters and accordingly there has been a shift in emphasis from water treatment to preventative measures including changes in land use management and the use of buffer zones (Burt et al., 1993). For soil erosion too, the main impacts lie off-farm, although the issue of reduced crop yields through loss of nutritious topsoil may soon become important. Flooding of property by soil-laden water has become a problem in many areas of north-western Europe (Boardman et al., 1994); houses and roads may be inundated, and river channels and lakes can become clogged with sediment (Heathwaite and Burt, 1992). Eroded soil carries pollutants such as phosphate (Sharpley et al., 1994) and pesticides (Harris and Forster, 1997); these can severely affect the health of aquatic ecosystems. There is significant erosion of peat soils in the uplands, where overgrazing has contributed to the problem; discoloration of water and sedimentation of reservoirs are particular problems in such areas (Burt et al., 1997).

In order to deliver environmental benefit, the movement of pollutants from farmland to river channels must be understood. This paper will consider the sources of pollutants (eroded soil, nutrients and pesticides), their transformation *en route* to the channel, and their eventual fate. (Further details of the source-pathway-target chain are given in Newson, 1992). Implications for catchment management are then addressed. Fundamental to all this is an understanding of the pathways by which runoff reaches the river channel, since sensitivity to change is closely related to the precise hydrological pathways operating.

2. Hydrological pathways in catchment systems

The nature of the soil and bedrock will determine the pathways by which hillslope runoff will reach a stream channel. It is clear from Fig. 1 that many different drainage routes exist. Some rainfall (or snowmelt) takes a fast track to the channel and contributes to stormflow; often this involves surface flow. Subsurface flow moves at much lower velocities, often by longer flow paths, and, although it may contribute to stormflow, its main effect is to maintain streamflow during dry periods through the sustained release of water from soil and bedrock. As Fig. 2 indicates, there are two types of storm runoff mechanism. When rainfall intensity exceeds the infiltration capacity of the soil, the excess begins to fill up surface depressions. Once these are full, excess rainfall overflows downslope and surface runoff begins; this is *infiltration-excess overland flow*

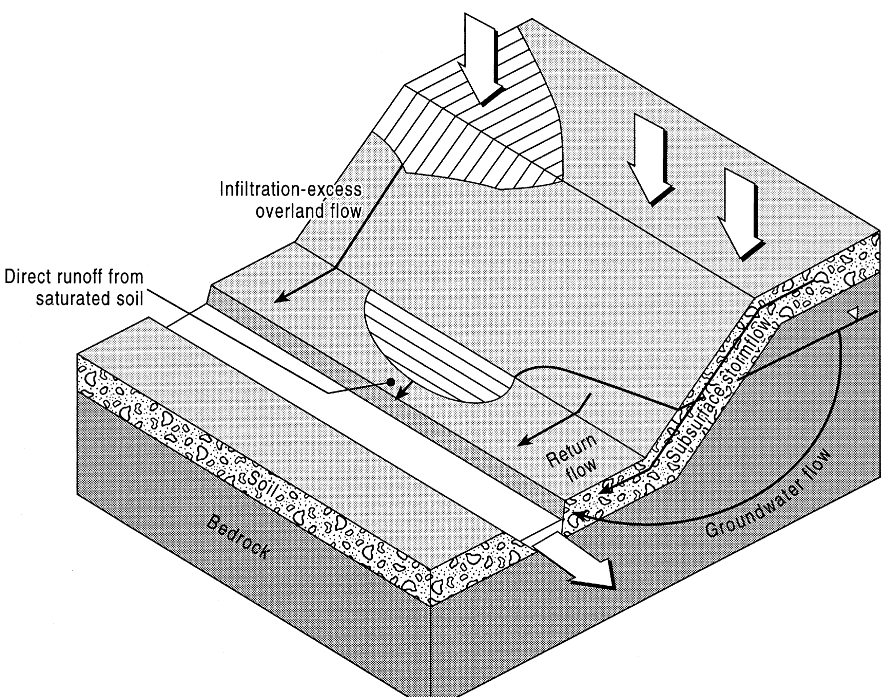

Fig. 1. Hydrological pathways. (1) infiltration-excess overland flow. (2) saturation-excess overland flow; (2a) direct runoff from saturated soil; (2b) return flow; (3) subsurface stormflow; (4), groundwater flow.

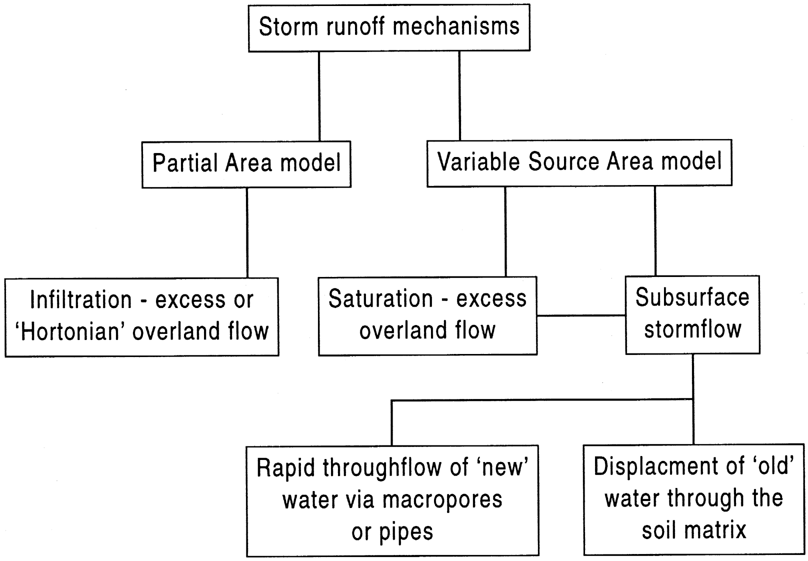

Fig. 2. Classification of storm runoff mechanisms.

(see Fig. 1(1)). Horton (1933) originally proposed that such runoff would be widespread across the catchment area, but Betson (1964) showed that infiltration capacity is not necessarily widespread even within a small basin. He proposed the *partial area model* of overland flow generation in which surface runoff is produced only from limited areas of the catchment in any given storm event. Despite its restricted extent, infiltration-excess overland flow can generate large flood peaks and is very often associated with soil erosion. Infiltration capacity is, therefore, one of the most sensitive hydrological variables. It used to be argued (e.g. Weyman, 1973) that infiltration-excess overland flow was rare in humid temperate regions, but it seems to have become much more common in some locations in the last two decades.

The *variable source area model*, first articulated by Hewlett (1961), recognises that subsurface flow and the production of saturation-excess overland flow are intimately linked. Where permeable soil overlies less permeable substrate, substantial quantities of subsurface flow (sometimes called 'throughflow' — Fig. 1(3)) may be generated. Soil water drainage tends to accumulate in favourable locations: at the foot of any slope, especially where the slope profile is concave; in areas of thinner soil where soil moisture storage is reduced; and, most importantly, in hillslope hollows where downslope drainage tends to converge. Where the soil profile becomes completely saturated, any excess water (further precipitation or additional downslope drainage — see Fig. 1(2a) and (2b), respectively) must flow over the soil surface. Such saturated areas usually develop in different locations to those producing infiltration-excess overland flow and it follows that other variables determine sensitivity to this mechanism of run-off generation.

Fig. 3 summarises the role of different variables in controlling storm runoff generation. Infiltration-excess overland flow is likely to occur in soils of low permeability

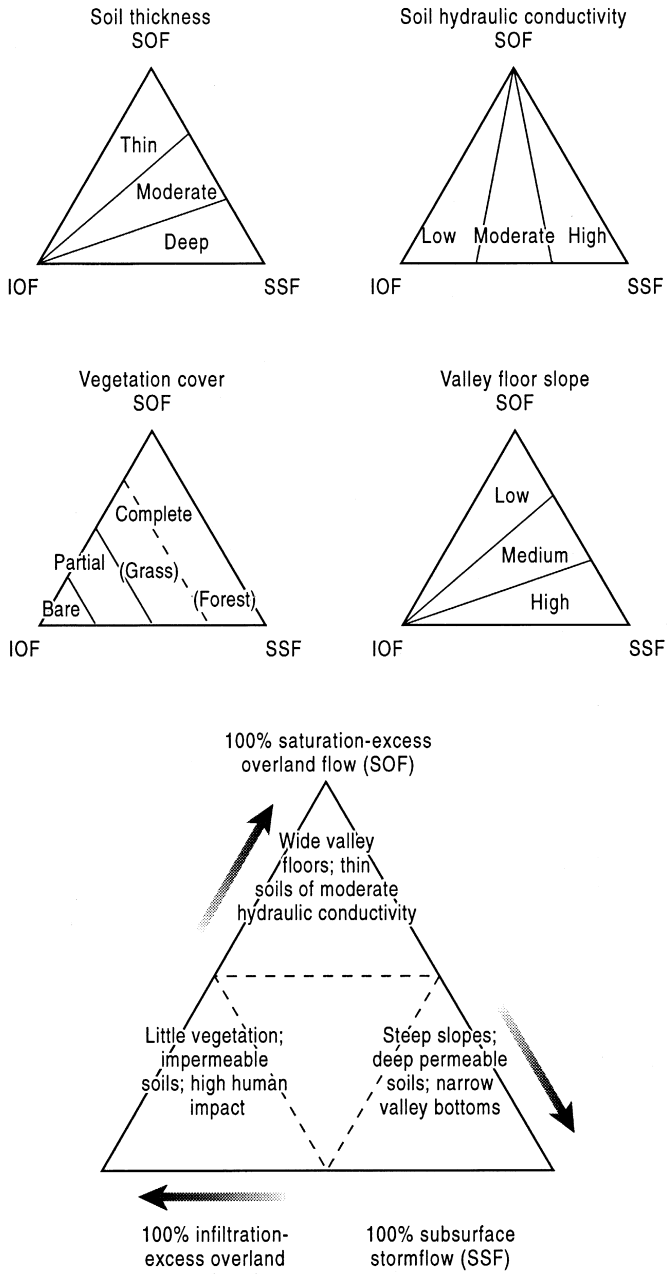

Fig. 3. The role of different controlling variables in storm runoff generation.

especially where there is little vegetation cover. Compaction of the soil surface, for example by heavy machinery or overgrazing, can cause this type of overland flow, even on usually permeable soils. Peak flow rates are high and runoff occurs quickly. At the other extreme, subsurface stormflow dominates where slopes are steep and soils are deep and permeable. Runoff rates tend to be low and there is a long delay between peak rainfall and peak runoff. Saturation-excess overland flow tends to occur in soils of modest permeability, especially where soils are thin and slope angles are low. Any changes that promote either type of surface runoff will increase the risk of flooding and soil erosion. On the other hand, an increase in subsurface flow, for example after installation of field drains, may well increase leaching losses. Further details of storm runoff production are given in Burt (1992).

Baseflow generation depends on the presence of an aquifer, a layer of rock or unconsolidated material that can yield significant quantities of water. Although the porosity of a rock determines how much water it can store, the specific yield, the amount of water released by drainage under gravity, is of more relevance in the supply of water to local rivers. This, together with the permeability and thickness of the rock (the product of these two is termed *transmissivity*) determines the importance of any aquifer (Burt, 1992). In the short term, all but the thinnest of aquifers are buffered from the effects of drought, but even a major aquifer like the Chalk of south-east England can be vulnerable to drought if the period of significant rainfall deficit lasts for several years (Burt and Shahgedanova, 1992). Changes in vegetation cover or soil surface condition can also affect the water balance of an aquifer in the long term. Traditionally, aquifers have been considered cheap sources of drinking water because of their isolation from surface pollution. However, surface pollution can eventually contaminate even deep aquifers, and once this happens, recovery (even if physically possible) will take at least as long (Burt and Trudgill, 1993).

Not all hillslope runoff flows directly into a stream channel. There is often between the slope and the stream — and not emphasised on Fig. 1 — a floodplain of significant width and minimal slope through which hillslope runoff must move. Given their structure and location, floodplains are likely to form wetlands: even when the floodplain is composed of permeable sediments, the water table will remain high, at least during the winter months. Floodplains may act both as a conduit and a barrier for hillslope water draining to the stream. The extent to which hillslope and channel are linked hydrologically depends on the pathway followed by hillslope runoff as it enters the floodplain zone. For example, overland flow may infiltrate a permeable floodplain, depositing its sediment load as it does so; in this case, a floodplain can provide an effective buffer zone between farmland and river. On the other hand, impermeable alluvium will cause slope drainage to be routed across the floodplain surface, reducing the residence time of water within the system and minimising the opportunity for pollution mitigation (Burt, 1997).

3. Definitions of sensitivity

Brunsden and Thornes (1979) expressed the sensitivity of a landscape to change as ‘the likelihood that a given change in the controls of a system will produce a sensible, recognisable and persistent response'. They defined landscape stability as a function of the temporal and spatial distributions of resisting and disturbing forces and invoked the factor-of-safety concept from civil engineering to describe the balance between barriers to change and the magnitude of disturbing forces. It follows that the sensitivity issue involves two aspects: the propensity for change and the capacity of the system to absorb the change. Assuming a constant distribution of disturbing forces, landscape sensitivity will therefore depend on either the capability of the system to prevent an impulse from having any effect (resistance) or its ability to return to its former state once disturbed (resilience).

Following Brunsden (1993), it is possible to identify four types of resistance (or barriers to change) within the landscape:

- Strength resistance describes the inherent ability of a material to respond to stress in a liquid, plastic or brittle way depending on circumstances.
- Morphological resistance recognises the importance of location, the (potential) energy of position, in relation to slope, relief and elevation.
- Locational resistance describes the proximity of sensitive elements within a system to the processes initiating change.
- Filter resistance describes the strength of coupling between individual elements of the system and the consequent ability of the system to transmit kinetic energy.

Variation in these resistances through time and space introduces a great deal of complexity into the way which earth surface systems respond to external forces. In terms of scale, it is clear that both the individual component and its position within the landscape must be considered. In terms of catchment management, this means that both single fields and their location within the drainage basin must be carefully thought about.

Brunsden and Thornes (1979) identify two end-members in the spatial response characteristics of a landscape to change.

• Mobile or labile systems have high sensitivity to external forces: they react quickly, relax to new system states with facility, and rapidly propagate effects downstream. These areas are morphologically complex because they are not only subject to rapid change and therefore exhibit transient forms, but (provided that low resistance is matched by high resilience) they are also capable of rapid restoration and the achievement of new stable states.

• Slowly responding or recalcitrant systems are insensitive to change: they have a ratio of stress to resistance that rarely exceeds unity. They lie far from the initial point of impact and changes only filter slowly through the system because of poor linkage, high storage capacity and intermediate buffering. Being insensitive to external effects, these areas change but slowly.

Landscape sensitivity and change has been reflected upon at some length by geomorphologists. The notions of resistance and resilience have also been incorporated into systems analysis in ecology (Holling, 1973; Odum, 1983). In catchment hydrology, however, such scrutiny has been lacking. What follows, therefore, is a preliminary attempt to examine the sensitivity of catchment systems to change. First, consideration is given to soil erosion, essentially a branch of fluvial geomorphology and so amenable to the sort of approach already described. Then the nitrate issue is examined: the interaction of nutrient cycling and hillslope hydrology must be analysed at the drainage basin scale.

4. Soil erosion and sediment delivery

Many British soils are now at risk of soil erosion. This is not a result of climate change but rather a lack of awareness by farmers and policy makers, and continuing financial incentives to grow arable crops on erosion-prone land (Boardman, 1990; Boardman et al., 1994). In the past only wind erosion was recognised in the UK but over the last two decades there has been a clearly identified increase in erosion by water, particularly on the intensively farmed lighter soils of the lowlands (Evans, 1990). The issue is not just about loss of topsoil; off-farm impacts are equally important and flooding of property by soil-laden water has become a contentious issue in many areas of north-western Europe (Boardman et al., 1994).

Farmers have been encouraged by generous subsidies to grow cereals on quite unsuitable land. Compaction by heavy machinery, loss of organic matter, a trend towards autumn-sown crops (leaving ground bare over the winter) and larger fields all contribute to the problem (Boardman, 1990). The erodibility of soils has thus increased or, to follow Brunsden (1993), the ‘strength resistance’ of soil to erosion by overland flow has become less. Some soils are, of course, more prone to erosion than others: sandy soils with a tendency to crust are perhaps most vulnerable, especially if heavy rain occurs when there is little crop cover — what Boardman (1992) describes as the ‘window of opportunity’ (Fig. 4). Once formed, crusts may endure through to harvest (Imeson and Kwaad, 1990) or may be broken up by frosts (Burt and Slattery, 1996). The tillage cycle of different crops and the vagaries of the weather make prediction of exactly which fields will erode in any given season difficult and complex. Autumn-sown cereals and spring-sown maize seem to be particularly problematic (see, for example: Heathwaite and Burt, 1992; Boardman et al., 1995; Burt and Slattery, 1996). Erosion of steep slopes (‘morphological resistance’), especially where these border the stream channel (‘locational resistance’), can be a major hazard. However, even fields remote from the channel can contribute sediment if there is good linkage provided by roads or tracks; such disjunct source areas can be as much of a problem as fields adjacent to the river. A common location for gully erosion is along the floor of dry valleys where surface runoff converges and accumulates; high discharge more than compensates for low slope (Slattery et al., 1994). Erosion control measures must focus on these locations as well as on the actual fields where the overland flow is generated. Once formed, such gullies may well persist in the landscape. Sensitivity to erosion may not correlate well with resilience. Thus, rapid recovery to the initial ungullied condition may not necessarily occur. This metastability illustrates that many systems do not behave linearly and that

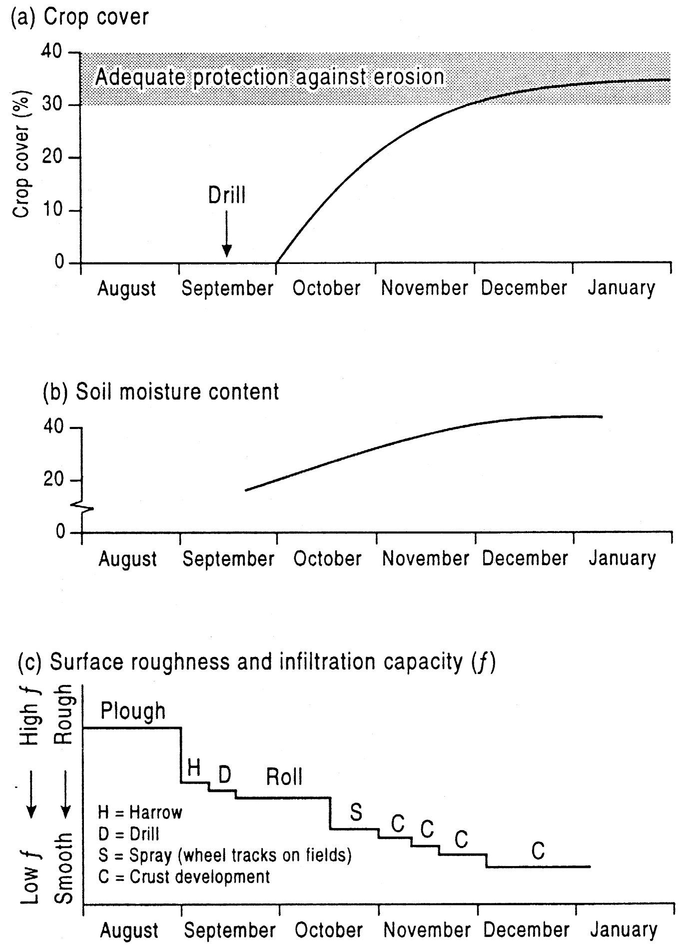

Fig. 4. Vulnerability of soil to erosion in relation to tillage and crop growth (after Boardman, 1992).

threshold change is an important element in any consideration of landscape sensitivity (Brunsden and Thornes, 1979).

Riparian buffer zones may function as sediment traps, reducing the coupling between slope and channel (i.e. increasing ‘filter resistance’). There is certainly evidence based on caesium inventories of slope-foot deposition of eroded soil, even without active management of the riparian zone (e.g. Walling, 1990), but it seems clear that active management of buffer zones would be needed to sustain their effectiveness in the long term (Dillaha and Inamdar, 1997). Riparian buffer zones may have most potential in higher order tributary valleys where a floodplain has formed; in low order tributaries steep slopes may border the channel and there is little opportunity for on-slope deposition of eroded soil unless barriers to overland flow like hedgerows are present. The sediment delivery issue is elaborated in Walling (1983, 1990).

5. Nitrate leaching in lowland catchments

One clear consequence of the intensification of agriculture since World War II has been the steady rise of nitrate concentrations in surface and ground waters (Fig. 5). This is a cause for concern with respect to human health and the stability of aquatic ecosystems, both freshwater and marine. The response shown in Fig. 5 may be interpreted as the result of several changes within the catchment. These include increased fertiliser use, increased frequency of ploughing, introduction of arable crops at the expense of grassland, reduction of permanent pasture in favour of temporary pasture, and higher stocking density on the grassland that remains. Thus, steady changes in land use and land management practices over many years have led to an increase in nitrate export from the catchment. The effect may well lag behind the cause: levels of soil organic matter take time to adjust to new conditions (Addiscott, 1988), and in areas with deep aquifers, it takes many years for the nitrate-rich leachate to percolate down through the unsaturated zone to the water table (Burt and Trudgill, 1993).

Despite recognition of the problem of rising nitrate concentrations, there has been little overt consideration of the sensitivity of catchments to nitrate leaching. One common method of modelling nitrate loss is the use of an export coefficient model. In its simplest form, this is a lumped, black-box model which calculates nitrogen losses from each area of cropland (including woodland) and each type of livestock (including people) found in the catchment (see Johnes, 1996, and Table 1 for further details). Despite its simplicity, the export coefficient model has proved very successful in modelling long-term nitrate loss from catchments (e.g. Johnes and Burt, 1993). Two elements of catchment sensitivity can be judged from Table 1. The export coefficients themselves indicate parameter sensitivity: for example, some crops involve higher losses than others so that a change from temporary grassland to cereals would in itself increase nitrogen losses, even without taking into account the increased frequency of ploughing which is likely to happen with arable cropping. Sensitivity of system state variables can also be assessed using a conventional sensitivity analysis: thus, a 10% increase in sheep numbers would have a more dramatic effect than a similar increase in pig numbers, given the absolute numbers of animals involved. However, such a simple model hides many aspects of system complexity that might usefully be discovered using a more sophisticated approach. Worrall and Burt (1999) modified an export coefficient model in an attempt to model soil nitrogen storage in a more dynamic manner. The release of soil nitrogen following ploughing of grassland was modelled as a first-order kinetic decay and the build up of reserves of soil nitrogen upon reversion to pasture was modelled both as a first-order and as a supply-limited process. Simulations indicated hysteresis caused by the supply-limited absorption of nitrogen to new pasture, suggesting that the ploughing up of permanent pasture is the dominant effect in controlling export from the catchment. The recovery of nitrogen reserves once grassland is re-established takes longer than the release. At the catchment scale, the model shows that nutrient export at any given time is related to the overall pattern of grassland management. Simultaneous ploughing of large parts of the catchment would, of course, have a dramatic effect, but in practice, ploughing in one place is usually balanced by reseeding in another. Currently the export coefficient model cannot easily deal with

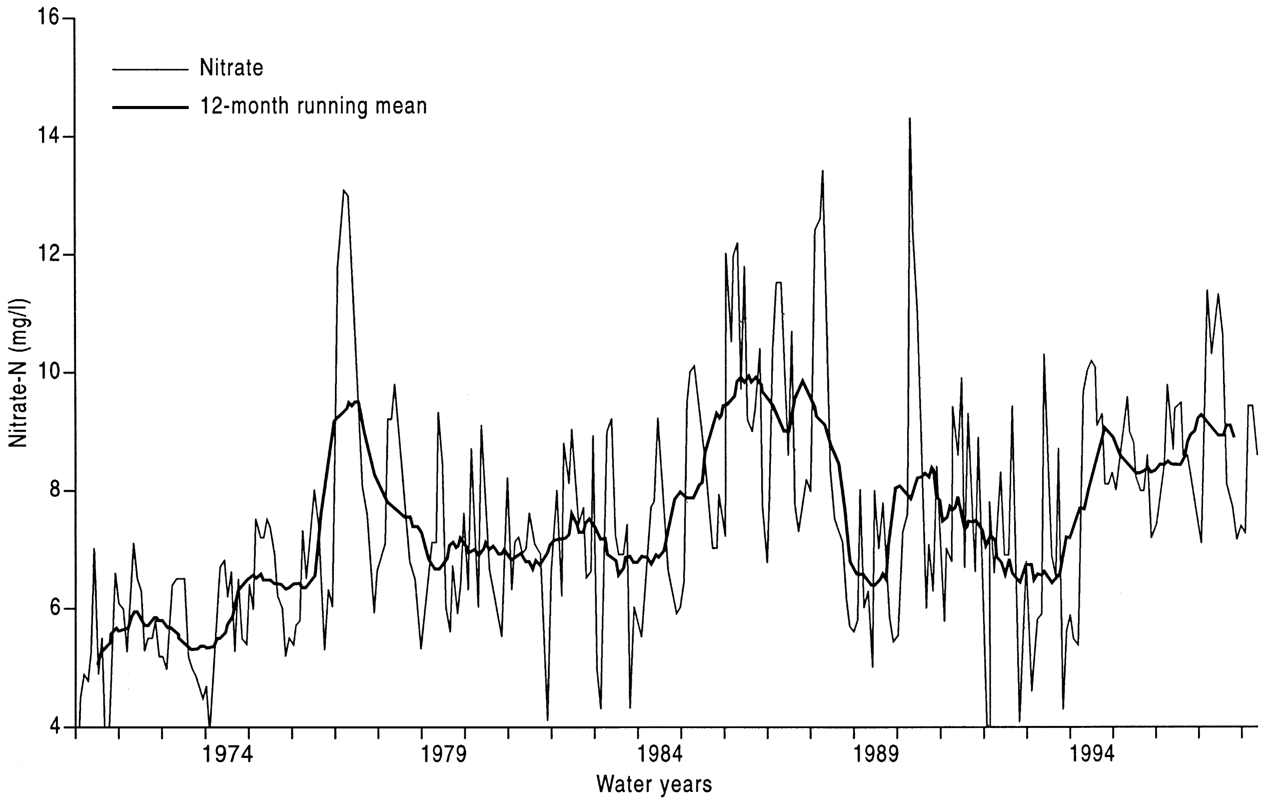

Fig. 5. The long-term nitrate record for the Slapton Wood catchment, Devon, England.

Table 1
Hypothetical annual nitrogen export for a small rural catchment estimated using an export coefficient approach

(1) Inorganic nitrogen losses from agricultural land

Land use	Area (ha)	Fertiliser application rate (kg ha^{-1})	Export coefficient	Loss (kg)
Temporary grass	20	200	0.05	200
Permanent grass	20	60	0.05	60
Cereals	30	170	0.20	1020
Other arable crops	10	160	0.30	480
Market gardening	4	60	0.20	48
Ploughed grassland	2	–	280 kg ha^{-1}	560
Total inorganic nitrogen loss from farmland				2368

(2) Organic nitrogen losses from agricultural land

Animal	Numbers	Annual loading per animal (kg)	Export coefficient	Loss (kg)
Cattle	50	70	0.15	525
Sheep	150	10	0.17	255
Pigs	20	20	0.12	48
Poultry	200	0.3	0.15	9
Total organic nitrogen loss from farmland				837

(3) Other losses of nitrogen form the catchment

• Atmospheric deposition at 5 kg ha^{-1} $\times$ 125 ha $\times$ 0.05(export coeff.)	31
• Human sewage at 50 $\times$ 4.5 kg (annual load) $\times$ 0.65 (export coeff.)	146
• Woodland at 39 ha $\times$ 10 kg ha^{-1} loss	390
Total 'other' losses	567
Total annual loss of nitrogen from catchment (kg)	3772
Catchment area (ha)	125
Total nitrogen loss per hectare per year (kg ha^{-1})	30.2

locational and filter resistance effects, although riparian functions can be included to some extent.

Physically based nitrogen cycling models have been developed to simulate cycling of carbon and nitrogen and to link this to crop growth and leaching (e.g. Whitmore and Addiscott, 1987). Such models can incorporate non-linearities and thresholds, something which export coefficient models cannot achieve. More importantly, catchment-scale applications are possible by coupling physically based soil nitrogen models to distributed hydrological models such as TOPMODEL (Whelan et al., 1995). There has as yet, however, been little use of such models to explore sensitivity issues (listed below).

• The identification of factors which render some catchments very sensitive to change. The scales of resistance identified by Brunsden (1993) are equally applicable here. Soil properties will influence nutrient cycling processes like mineralisation and denitrification, and so control the source of nutrients for leaching. Catchment topography ("morphological resistance") will affect the efficiency of nutrient export from the catchment system by influencing the pathways by which nutrients reach the river (e.g.

Burt and Arkell, 1987). Location plays a role too: fields near the channel may have more influence than distant ones, but the nature of coupling along flow lines will also be influential (“filter resistance”).

• The capacity of a catchment system to absorb change, for example, the importance of specific locations within the catchment. There has, for example, been much interest in the possible role of riparian buffer zones providing a barrier between farmland and the river (Haycock and Burt, 1993). Conditions which favour denitrification — nitrate-rich water flowing through permeable, carbon-rich sediments — are found on some floodplains and this may provide a ‘last line of defence’, given that the leaching has already taken place. Again, some locations may be more sensitive than others and change at these points (e.g. under-drainage of floodplains) may have a disproportionately large effect.

•The amount of change taking place at any one time. All catchments are subject to land use change, although this is often at a slow rate, only a few fields changing each year. However, if a large number of farmers change their practices simultaneously, for example in response to a new crop subsidy, this can have a large impact in the short-term (Worrall and Burt, forthcoming).

• The rate at which change takes place in different catchments and their resilience to recover after disturbance. This matter has not been modelled although a good deal of empirical evidence exists against which model output might be compared (e.g. Betton et al., 1991).

• The importance of hydrological pathways in determining sensitivity to nitrate leaching and the possibilities of managing these to minimise nutrient export (This is essentially a combination of Brunsden’s “locational” and “filter” resistances.). Recent interest in nitrate buffer zones is again relevant here. It seems that buffer zones will function most efficiently where hillslope water flows through the floodplain sediments (rather than bypassing the floodplain system). It may prove possible in some situations to manage water table elevation so as to optimise denitrification (Burt et al., 1999).

• Quantification of risk. Worrall (pers. comm.) notes that risk to groundwater systems can be evaluated as the product of vulnerability (relates to innate properties of the aquifer) and hazard (relates to actions taken within the aquifer catchment). It would be interesting to apply such ideas to surface water systems.

6. Towards integrated management of water quality in rural catchments

The impact of modern farming on rural catchments is seen in two ways: on and off the farm. On the farm, concerns relate to soil degradation; off-farm impacts include water pollution, sedimentation and flooding. Following Brunsden’s notion of resistance, we can identify three scales at which management can help conserve soil quality and mitigate pollution: soil resistance (in-field), locational resistance (fields within a single farm) and filter resistance (linkage between fields along a flow line).

On-farm strategies have focussed on preventative measures, but this is rarely an attractive option for farmers unless there is appropriate financial compensation in return for significant changes in land use and management practices. Prevention of nitrate leaching at source has been approached through a combination of good farming practice and more elaborate schemes (e.g. The Nitrate Sensitive Area (NSA) scheme) which offer financial compensation in return for significant changes in land use and management practices (Burt and Haycock, 1993). Lord et al. (1999) show that significant reductions in nitrate leaching were achieved within the NSA scheme. However, they were unable to present any direct evidence of reduced nitrate concentrations in groundwater, only the results of soil water sampling and model calculations. It remains unproven therefore whether local protection measures like the NSA scheme can be successful. In relation to erosion, soil conservation schemes remain voluntary and there is no incentive for farmers to target erosion-prone sites. In either case, of course, the farm remains the unit of planning whereas an appropriate hydrological plan demands appreciation of runoff pathways and the delivery of water, nutrient and sediment to the river channel. Nevertheless, *whole-farm planning* can achieve a certain level of success by focussing soil conservation practices on the most vulnerable locations. It may be that in the future there will be cross compliance between subsidy payments and pollution prevention measures. Through production of whole farm plans, farmers will be able to target set-aside and other practices, avoiding the blanket application of measures that has hitherto been imposed.

In Australia and New Zealand the *Landcare* movement has shown that an integrated approach to water quality management and other planning issues can be achieved at the catchment scale, with farmers and other interested parties coming together in a spirit of co-operation (Burt, 1999). Whole-farm planning may achieve a certain amount in terms of pollution prevention and soil conservation, but without concerted *whole-catchment* planning (often called "total catchment planning" in Australia), it will be difficult to achieve a consistent and widespread level of improvement. At the catchment scale, some zones may be particularly sensitive and will therefore require maximum protection, whilst others areas may be much less vulnerable. With integrated catchment planning, based on a detailed understanding of landscape sensitivity, environmental protection measures (and the accompanying compensation) can be carefully targetted. This may mean, of course, restrictions (and payments) for some farmers and none for others, depending on which actions are needed in which part(s) of the catchment. Whether such an approach is acceptable remains to be seen.

Agricultural policies of successive governments (both UK and EU) have been a major cause of the countryside being overexploited by intensive farming. Whether or not the objectives have been met (these include increased production, a fair standard of living for the agricultural community, and available supplies at reasonable prices), it is clear that the cost, including environmental degradation, has been high (Whitby et al., 1996). In the future, we may anticipate cross-compliance between financial support for farming and the need to maximise environmental protection. Soil conservation in the widest sense must become an important element of such policies as farmers seek to sustain soil productivity and minimise off-farm impacts. In this way, productive, sustainable farming may co-exist alongside other rural land uses, providing clean water, better access, improved habitats and a diverse attractive landscape. To achieve this end, farming and water supply interests must work together, not an easy matter in countries where such responsibilities are split between separate government ministries.

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