The Ninth Ineson Lecture

Assessing and Controlling the Impacts of Agriculture on Groundwater—from Barley Barons to Beef Bans

S. S. D. Foster

Assistant Director, British Geological Survey, Nottingham, NG12 5GG & Wallingford OX10 8BB, UK.
Visiting Professor of Contaminant Hydrogeology, University of London, Royal Holloway College

Abstract

One of the less obvious, but most insidious, effects of the enormous changes in English agriculture over the past 50 years has been the contamination of groundwater—originally by nutrients, subsequently by pesticide residues, and most recently by the livestock parasite Cryptosporidium. The paper analyses scientific understanding of the factors determining the origin, transport and attenuation of these contaminants in groundwater systems. It focuses primarily on the British situation, but draws on experience from Europe and beyond as appropriate. Given the essentially diffuse character of the mainly agricultural activity generating the subsurface contaminant load, pollution control has presented (and continues to present) a complex regulatory challenge in legal, technical, economic and social terms. The discussion explores this challenge, which yet again has become very topical at national and European policy levels.

Keywords: environmental impact, groundwater contamination, land use, pesticides, unsaturated zone.

Introductory remarks

Scope and objective of paper

This paper has a number of objectives:

- first, to summarize the state of scientific knowledge on the input, attenuation and transport of the main groups of agriculturally-derived contaminants which threaten groundwater quality
- second, to review the status of policy responses on controlling the threat of groundwater pollution from such contaminants
- third, to look forward (over the next 25 years) and identify potential future threats to groundwater quality, arising from agricultural activity.

The current status of diffuse agricultural pollution of groundwater is set in a historical context by looking back over the past 25 year's experience. The paper begins with a brief introduction to the emerging groundwater nitrate problem of the mid-1970s, which turned out to be perhaps the major European groundwater quality concern of the 20th century. Important advances in hydro-scientific concepts have been achieved incidentally as a result of the pursuit of research on diffuse agricultural pollution issues over the same period, and these are also highlighted in the paper.

This is followed by an assessment of recent experience in relation to groundwater contamination by pesticide residues and the livestock parasite cryptosporidium, then returning to more recent work in relation to nitrate soil leaching and subsurface attenuation capacity. The approach used for discussion in each case is to consider in sequence knowledge of the input, attenuation and transport (as represented schematically in Fig. 1) of the respective contaminants in the major British aquifers. It should be pointed out from the outset that groundwater pollution from all these contaminant groups can also arise from non-agricultural (essentially urban) sources, but this aspect is not discussed in detail in this paper. Moreover, the effects of the application of sewage sludge to agricultural land are also not discussed here, although when such application conforms in terms of timing and loading with the standard guidelines, any hazard to groundwater should be restricted to certain pathogens and synthetic organic compounds.

From the paper's subtitle it will be evident that it goes beyond the scientific understanding of the subject in hand and enters into its political, economic and social dimensions, which is a new departure for the Ineson Lecture Series.

The paper focuses geographically upon the extensive outcrop areas of the major British aquifers in lowland England (the Chalk, the Permo–Triassic Sandstones and the Jurassic Limestones), since this region is also subjected to intensive agricultural cultivation. However, it should be of relevance to most parts of Britain and also to many neighbouring areas of mainland Europe. It has, however, to be recognized that it may not be appropriate to extrapolate conclusions from these regions to those where more-or-less continuous irrigation is required to sustain agricultural cultivation.

It should also be recognized that the problem of diffuse agricultural pollution of groundwater applies equally to bottled mineral waters as to public water-supplies (except where the former are proven to be of pre-industrial age or from non-farmed catchments), and is even more acute in many rural private supplies.

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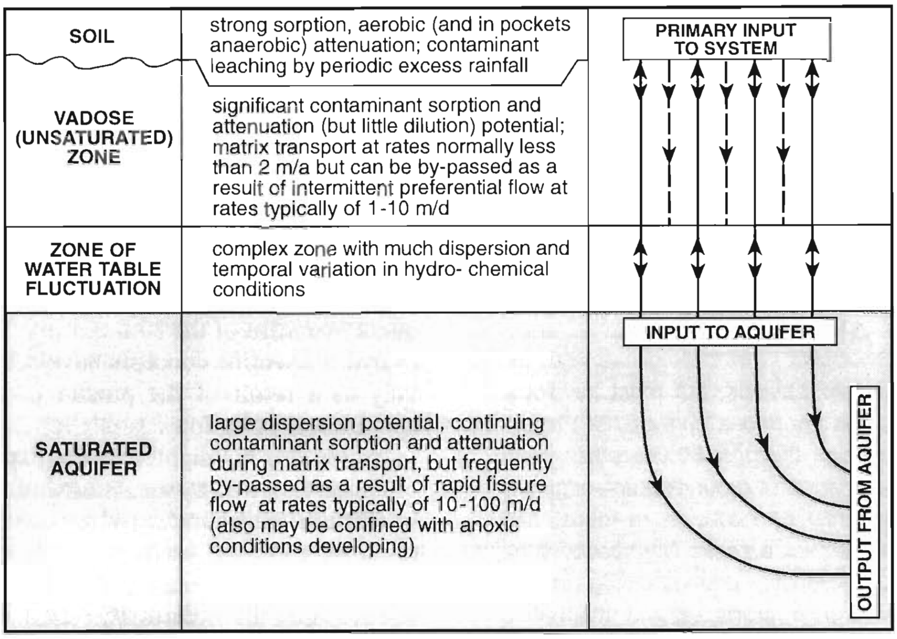


SOIL	strong sorption, aerobic (and in pockets anaerobic) attenuation; contaminant leaching by periodic excess rainfall
VADOSE (UNSATURATED) ZONE	significant contaminant sorption and attenuation (but little dilution) potential; matrix transport at rates normally less than 2 m/a but can be by-passed as a result of intermittent preferential flow at rates typically of 1-10 m/d
ZONE OF WATER TABLE FLUCTUATION	complex zone with much dispersion and temporal variation in hydro- chemical conditions
SATURATED AQUIFER	large dispersion potential, continuing contaminant sorption and attenuation during matrix transport, but frequently by-passed as a result of rapid fissure flow at rates typically of 10-100 m/d (also may be confined with anoxic conditions developing)

Fig. 1. Generalized compartmental scheme of groundwater flow and pollutant attenuation.

Emergence of the groundwater nitrate problem

Increasing concentrations of nitrate in groundwater supplies drawn from the major unconfined aquifers of eastern England was first recognized on a fairly widespread geographical basis in the 1970s (Foster & Crease 1974; Foster 1976). Background levels beneath the few remaining areas of natural vegetation are not believed to exceed 1 $mgNO_3/l$. The increases were to levels approaching or in excess of 50 $mgNO_3/l$ and appeared to be a side-effect of major changes in agricultural cultivation during 1950–70, which were directed towards (largely successful) efforts to increase national grain production. These changes had a number of distinct components:

- the conversion of extensive areas of permanent pasture to tilled land under arable cultivation
- major increase in the total proportion of agricultural land dedicated to cereal cultivation, in many areas from less than 25% to more than 50% during the period 1950–70
- substantial increase in the application of inorganic fertilizers to sustain more continuous cereal cultivation, with rates widely increasing from less than 50 kgN/ha/a to more than 100 kgN/ha/a.

The overall effect of these changes, coupled with some intensification of grassland-based agricultural production, is summarized in Table 1. It will be evident that the 20-fold increase in applied fertilizer nitrogen during 1940–80 achieved a 3-fold increase in food production, with the implication that the remainder (more than 1000 kT/a) remained in soils, from where it was lost to the atmosphere by denitrification, or leached to surface and groundwater as nitrate.

Table 1. *Estimate of the overall British agricultural nitrogen balance for 1940 and 1980 (Gasser 1982, discussion)*

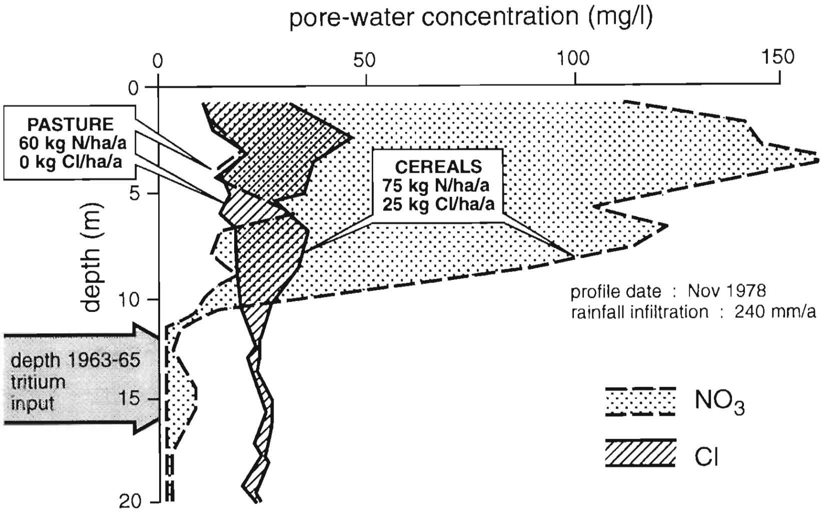
Year	N application in fertiliser (kT/a)	N content of home-produced food (kT/a)
1940	60	43
1980	1270	147

The other critical issue to emerge in the 1970s was the significance of the vadose zone in diffuse pollution, and the potentially very slow average rates of downward transport of soluble contaminants (like nitrate) leached from cultivated soil, even in the fractured porous aquifers characteristic of much of lowland England (Foster & Crease 1974; Young *et al.* 1976; Foster & Young 1980). This mode of vadose zone transport was suggested through profiling of environmental tritium in pore waters (Smith & Richards 1972) and subsequently confirmed by observing long-term temporal changes in vadose zone nitrate profiles themselves (Geake & Foster 1989; Kinniburgh *et al.* 1999). It introduced the possibility of a time-lag of decades between the onset of agricultural land-use changes and their impact being felt in the quality of groundwater drawn from deep aquifers.

Advances in hydrogeoscience through pursuit of diffuse pollution: progress over 25 years

Correlation between groundwater quality and land-use in unconfined aquifers

Prior to the 1970s there was no appreciation of the correlation between groundwater quality in phreatic



envelopes represent increased leaching consequent upon ploughing part of an unfertilized pasture in 1966 and subsequent cultivation of cereals; the other part remained as low productivity pasture

average applications of N and Cl (as KCl) in fertilizers during 1967-78 are indicated for each case

Fig. 2. Vadose zone profiles from the East Yorkshire Chalk showing the effect of converting pastureland to cereal cultivation (Foster *et al.* 1982).

aquifers and the nature of agricultural practices on the overlying land. At that time, British groundwater research was dominated by efforts to evaluate the hydraulic structure of the major aquifers—especially the Chalk—and its significance for direct water-supply development and the exploitation of groundwater storage for drought river-flow augmentation. Little attention was paid to groundwater quality issues and it was widely assumed that the main control on groundwater chemistry was the interaction between infiltrating rainfall and the aquifer materials themselves.

The introduction of vadose zone water-chemistry profiling in the 1970s revealed remarkable contrasts between the quality of groundwater infiltrating almost identical soil profiles other than their cultivation (lightly-fertilized pasture and arable cultivation respectively), not just in relation to nitrate but also chloride (Fig. 2) and a range of trace elements. Prior to the 1970s, ammonium sulphate was the predominant form of inorganic N fertilizer, and this, together with uncontrolled emissions of sulphur dioxide from thermal power stations led to elevated SO_4 concentrations in groundwater recharge (Foster 1989), which reduced markedly except where applied to specific crops.

Vadose zone contaminant transport in fractured media

Until the 1970s it was assumed that the transport of all aqueous solutes and contaminants through the vadose zone of fractured aquifer media was a rapid process. The introduction of environmental tritium profiling for vadose zone pore waters challenged this interpretation and demonstrated the potential for slow rates of downward transport (Smith & Richards 1972; Foster & Crease 1974).

The potential for exchange through aqueous diffusion between dissolved chemical species flowing in fractures and the surrounding aquifer matrix was first recognized by Foster (1975), and subsequently was the subject of intensive investigation by a large number of researchers (reviewed by Pankow & Cherry 1996). The diffusive mechanism was found to be of major significance, because of the small distance separating fracture water from matrix pore-water, and the initially steep concentration gradient (Fig. 3). The dominant influences over this transport process are fracture aperture and spacing, matrix porosity and tortuosity and the corresponding aqueous diffusion coefficient (Barker & Foster 1981), which varies widely between small (dissolved) ionic species and larger organic molecules or microbiological contaminants.

Although this mechanism reduces aquifer vulnerability to pollution by degradable soluble contaminants, it merely introduces a substantial time-lag for transport of persistent contaminants. The definition of this transport mechanism has also had a major bearing on the interpretation of contaminant transport in fractured and heterogenous media in a wide variety of applications. Many variants have been considered, including the effects of mineral coatings and biological films on fracture walls.

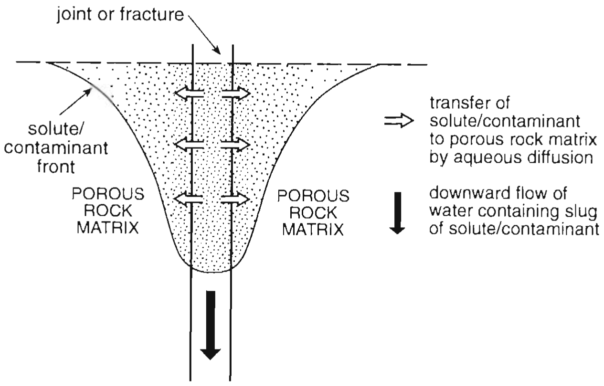


Fig. 3. Conceptual scheme of groundwater and solute transport in a fractured porous media with solute exchange by aqueous diffusion.

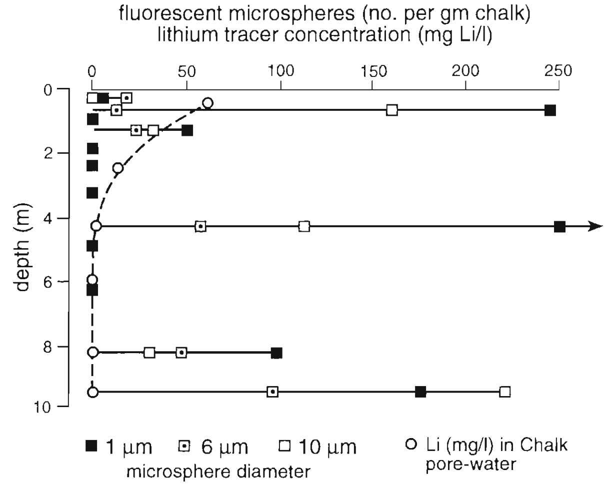


Fig. 4. Preferential flow in a Chalk vadose zone demonstrated by differential movement of lithium bromide and fluorescent polystyrene microspheres.

Scale and significance of preferential vadose zone flow

Preferential flow is a term taken to include all forms of rapid downward water movement in the vadose zone through fissures and macropores, effectively by-passing the porous matrix. It is likely to be highly intermittent and can occur in profiles that at other times support a predominantly slow downward movement of solutes and contaminants through diffusive exchange (Beran & Germann 1982; White 1985). However, the potential for preferential flow is of major importance in terms of increasing the vulnerability of underlying aquifers to contamination by agricultural land-use practices.

The hydraulic properties of most British aquifers (with fracture-related hydraulic conductivity) are such as to present high probability of intermittent vadose zone preferential flow. It can be initiated by various factors, and is often associated with instability of downward flow in situations where a permeable horizon is overlain by a less permeable one.

Preferential flow, however, is difficult to prove (especially under natural rainfall conditions) and because of the 'pressure to model' has almost certainly been underestimated in past numerical models of nitrate transport in the vadose zone. Its occurrence complicates considerably the use of vadose zone profiles to predict future trends in nitrate concentration in groundwater sources (Geake & Foster 1989).

Indirect evidence for its existence comes from various lines of circumstantial evidence including:

- intermittent groundwater contamination with faecal bacteria in areas of deep (10–25 m) water table, despite the fact the profiles of environmental tritium indicate average rates of vadose zone transport for non-reactive solutes of less than 1.0 m/a
- the fact that it is not possible to reach satisfactory mass balances for environmental tritium profiles in the vadose zone, even at locations where the slow flow component does not show significant dispersion effects and revised soil moisture models are used (Foster & Smith-Carington 1980)
- the substantial number of vadose zone pore-water profiles for environmental tritium which show marked dispersion with strong forward tailing of the 1963–65 peak concentration; this could only occur as a result of fissure flow in a dual-porosity system where the velocity of water movement was high enough to prevent full interchange of solutes with the microporous matrix (Geake & Foster 1989).

Preferential flow will always be characterized by much more rapid contaminant transport, providing less opportunity for retardation through molecular diffusion into the microporous matrix and associated adsorption, chemical reaction and biodegradation (Barker & Foster 1981). If preferential flow in fissures of large aperture occurs, the possibility of transport of less mobile contaminants adsorbed on colloidal material is also possible (McDowell-Boyer *et al.* 1986).

Various novel approaches are being introduced in attempts to understand further the significance and scale of recharge by preferential flow:

- shallow-depth/high-resolution geophysical surveys with modern data inversion technology (possibly combined with tracer techniques) offer potential for mapping the subsurface distribution of potential preferential flow horizons (Andrews *et al.* 1995)
- use of a cocktail of soil injected tracers (lithium bromide, a bacteriophage and fluorescent polystyrene microspheres) demonstrated that a Chalk vadose zone of 10 m thickness was traversed in 0.6–3.0 days following 'light irrigation' (Fig. 4), with the fluorescent microspheres being detected on internal fractures in drill cores
- the application of the stable isotope ^{2}H and a chloride tracer over a period of 4 years at an Upper Chalk site

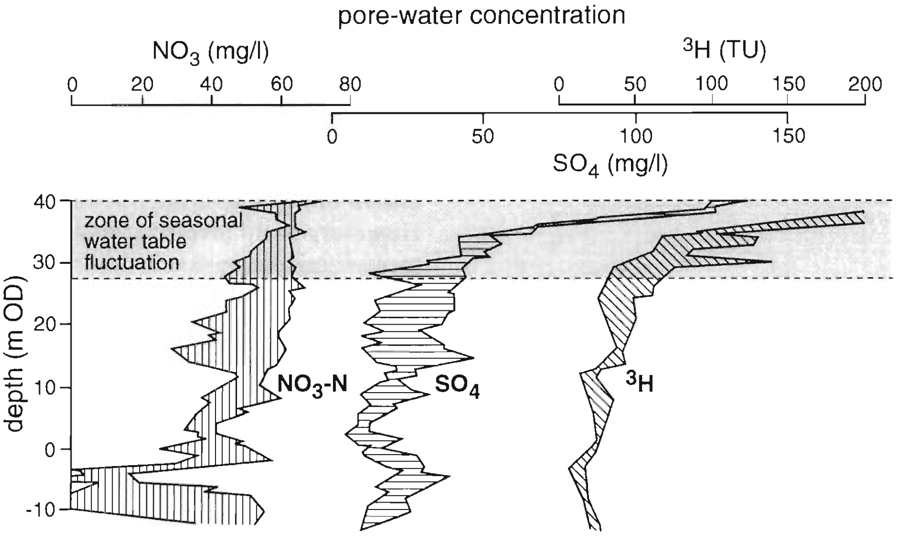


Fig. 5. Profiles of saturated zone pore-water chemistry in the Norfolk Chalk aquifer showing marked vertical stratification (Sep 1976 data).

Table 2. *Variation of groundwater quality with depth below water-table in an arable farming catchment on the unconfirmed Triassic Sandstone aquifer in South Yorkshire (Nov 1980 data)*

Depth below Water-Table (m)	Groundwater Chemistry (mg/l)						
	NO_3	Ca	Mg	Na	K	Ba	B
0–10	50–150	>200	>50	>30	>6	>0.4	>0.6
10–40	40–60	90	30	20	5	0.3	0.2
40–80	20–40	40	10	10	2	0.2	0.1

on the other hand revealed little non-dispersive flow, although only about 50% of the total porosity appeared to be active in solute transport (Barraclough *et al.* 1994).

What has now become clear, from the limited evidence available, is that some areas of the major British aquifers present a high probability of preferential vadose zone flow, and that localized parts of their outcrop area could be defined as having extreme vulnerability in view of the presence of karst-like solution features.

Saturated zone quality layering

In the 1970s little was known about vertical variations in groundwater quality within aquifers, and the main variations in groundwater chemistry were assumed to occur areally. Investigations into the degree of penetration of agricultural contaminants into groundwater systems revealed very widespread water-quality layering in phreatic aquifers (Fig. 5) (Table 2).

The significance of this for groundwater resource evaluation and monitoring practice has become increasingly apparent and today more effort is generally put into evaluating the significance of such variations. They normally arise as a result of the hydraulic structure (and generally higher horizontal conductivity) of aquifer systems, together with temporal deterioration of groundwater recharge quality as a result of diffuse pollution. It should be noted that for fractured aquifers, similar dual-porosity contaminant and solute transport processes occur in the saturated zone with exchange of recent contaminants between fracture water and porous matrix, although full exchange will not take place in those aquifers characterized by high fissure-flow velocities.

Subsurface microbial activity

In 1980 it was said that less was known about natural microbial activity a few metres below the Earth's surface than was known about microbial life on the surface of the Moon! Information on naturally-occurring micro-organisms in aquifers was very scant, in part because early studies suggested that the numbers of bacteria dropped dramatically with increasing depth. It was, therefore, widely assumed that such organisms were absent, other than in rather special circumstances, such as small zones characterized by sulphate reduction.

However, it has become apparent, in part through the investigation of diffuse groundwater pollution, that

Table 3. *Typical regime of pesticide application to an individual arable field in lowland England*

Year	Crop	No of Treatments	No of Compounds
1987	Spring Barley	4	5
1988	Winter Barley	5	6
1989	Sugar Beet	7	9
Total		16	20

microorganisms are quite widely distributed to considerable depth and could be involved in a number of important hydrogeochemical processes (Whitelaw & Rees 1980; Foster *et al.* 1985; Balkwill & Ghiorse 1985), even in areas not experiencing a heavy subsoil loading of dissolved organic carbon from anthropogenic pollution sources. However, more detailed data on their distribution and rates of *in-situ* metabolic activity still remain very scant.

Groundwater pesticide contamination: the more you look the more you find!

Facets of assessment problem

Prior to the late 1980s there was not much concern about the possibility of groundwater pollution by pesticide residues. Agricultural scientists argued that soil sorption of the higher molecular weight compounds (such as the chlorinated hydrocarbon insecticides) and volatilization of lower molecular weight compounds (like most herbicides) would predominate. In fertile soils most pesticide compounds in current use in Britain have half lives of less than 100 days, and thus it was assumed that soil residues would be eliminated (by aerobic biodegradation or chemical hydrolysis), preventing groundwater contamination except where the water-table was shallow, soils coarse-grained and pesticide applications excessive.

To evaluate pesticide leaching to groundwater, data are needed on the 3-dimensional subsurface distribution of pesticide compounds in aquifer recharge areas. However, such investigations are proving costly and problematic because of:

- the wide range of pesticide compounds in common use, even at the level of an individual field (Table 3)
- the need to work at very low concentrations because of the high toxicity of many compounds, necessitating large sample volumes and careful handling to avoid compound modification and volatile loss.

In view of these difficulties, an essential prerequisite for risk assessment is to identify the most likely sources of pesticide contamination and the most probable mechanism of transport from the land surface to groundwater. Meaningful evaluation can be undertaken using the published biochemical properties of pesticides themselves, the characteristics of soil-aquifer systems involved, and the local rainfall regime and irrigation practices. Even simplified empirical formulae based on some of these factors can be useful in assessing the likelihood of leaching to groundwater (Table 4).

Factors affecting leaching to groundwater

Rates of agricultural pesticide application in Britain are generally in the range 0.2–10.0 kg/ha/a of active ingredient, thus leaching losses of 1% or less could be significant in relation to the EC drinking water guideline (0.1 μg/l). Many compounds have water solubilities in excess of 10 mg/l (10 000 μg/l) and this is not a limiting factor in leaching from soils. Moreover, high pesticide concentrations have now been observed in soil-water

Table 4. *Empirical indices of groundwater contamination hazard based on susceptibility of pesticide compound to soil leaching*

Index (author) formula	Pesticide Properties			Soil Properties			
	Koc (cc/g)	$T_{\frac{1}{2}}$ (d)	others —	i (cm/hr)	z (m)	fom (—)	others (—)
Groundwater Ubiquity Score (GUS) $GUS = \log_{10} (T_{\frac{1}{2}}) [4 - \log_{10} Koc]$	×	×	—	—	—	—	—
Leachability Index (LI) $LI = (S.T_{\frac{1}{2}})/(V_p.Koc)$	×	×	S (mg/l) V_p (mPa)	—	—	—	—
Retardation Coefficient (R)/Transit Time (T_c) (Jury *et al.* 1987) $R = 1 + (Koc.fom.\theta.\rho B)/T_c = z\theta R/i$	×	—	—	×	×	×	ρB(g/cc) θ (—)
Attenuation Factor (AF) (Rao *et al.* 1985) $AF = \exp [-0.69\ T_c/T_{\frac{1}{2}}]$	×	×	—	×	×	×	ρB(g/cc) θ (—)

Koc, pesticide organic-carbon partition coefficient; $T_{\frac{1}{2}}$, pesticide soil half-life; S, pesticide water solubility; V_p, pesticide partial vapour pressure; fom, soil organic matter fraction; θ, soil volumetric moisture content; i, soil infiltration capacity; z, soil thickness; ρB, soil dry density.

solution and overland flow, often exceeding 10 μg/l for up to 50 days after application and reaching 1000 μg/l for 10 or more days after application.

Of greatest significance in the leaching from soils is the mobility of the pesticide compound in soil solution. This will vary with the soil organic matter and/or clay mineral content, and can be expressed by the corresponding partition coefficient. Both the modes of application and action of the pesticide are also important, since most compounds targeted at plant roots and soil insects have to be much more mobile than those acting directly on plant vegetation (Foster *et al.* 1991). Another important factor is the volatility of the pesticide compounds, since losses from the soil via the vapour phase are often very significant, and a similar process could continue deeper in the vadose zone.

The degradability of pesticide compounds in the soil horizon, as a result of bacteriological oxidation or chemical hydrolysis, will normally be significant. Soil half-lives for compounds in widespread use in Britain can range from 10 days to 10 years, but for the more mobile compounds are normally less than 100 days. However, given the timing of applications, even the latter are sufficiently persistent to remain in the soil during periods when leaching may occur. Moreover, some derivatives of partial oxidation or hydrolysis may be more toxic and/or mobile than the original parent compound.

It must be borne in mind that the adsorption and degradation characteristics of pesticides listed in handbooks of commercial chemicals (in terms of partition coefficient and half-life) relate to a fertile organic clayey soil. They may not be representative of (more permeable) sandy soils widely developed on aquifer outcrops, which are likely to be more vulnerable to pesticide leaching, and even less so of the aquifers themselves (Foster *et al.* 1991).

Subsurface transport and fate of pesticide residues

Pesticide compounds leached from permeable soils enter an environment, which contains a much smaller proportion of clay minerals and organic matter, and a greatly reduced population of indigenous bacteria. Thus the mobility and persistence of all pesticide compounds should be expected to be many times greater in the groundwater system than in a typical agricultural soil (Table 5). However, it is unlikely that downward transport rates in the vadose zone matrix exceed 1 m/a (and for many compounds they will be much slower). Since few compounds have been in regular use for more than 10–20 years, most pesticides leached from agricultural soils would be expected still to remain in the vadose zone and/or to have been eliminated by biodegradation or volatilization, if this was the only transport mechanism operative.

Table 5. *Reduction in the half-life of commonly-used pesticide compounds in groundwater compared to fertile soil (mean of 3 individual batch tests; for groundwater tests with soil phase was included and pH=7)*

Pesticide Compound	Half-Life (d)		Ratio
	Fertile soil	Groundwater	(gw/soil)
2.4B	10	1200	120
Alachlor	20	1200	60
Dichlorprop	10	900	90
Propanil	3	240	80

The hydraulic characteristics of most British aquifers are, however, such as to present high probability of preferential vadose zone flow, which is of major importance in the consideration of pesticide contamination hazard for groundwater. Where developed, it would provide routes for deeper penetration of pesticide compounds readily leached from the soil by winter rainfall. These routes would be characterized by much more rapid pollutant transport providing less opportunity for retardation through molecular diffusion in the microporous matrix and subsequent adsorption, reaction and biodegradation. The stringent EC directive for pesticides in drinking water (0.1 μg/l) has already been exceeded in numerous British public water-supply boreholes, although concentrations in excess of 1.0 μg/l have seldom been recorded. The pesticides most frequently encountered to date are all herbicides—atrazine, simazine, mecoprop and isoproturon. Detailed monitoring of the Chalk water-table at sites in Hampshire under near-continuous arable cultivation using regular applications of isoproturon (to winter cereals) or atrazine (to maize) has revealed only limited penetration of these pesticides into groundwater. Most transport was via preferential flow, as revealed immediately after the onset of recharge through use of a floating water-table sampling device (Fig. 6).

It is most important to record that at present there has been little research or monitoring for the so-called metabolites (partial breakdown products) of pesticides. A recent and systematic survey of pesticide occurrence in the phreatic aquifers of the maize and soyabean region of mid-continental USA detected 27 pesticide compounds in 810 groundwater samples (Kolpin & Goolsby 1995). Of the six most widely-detected compounds no fewer than five were herbicide metabolites, and amongst these the presence of alachlor derivatives was especially noteworthy since the parent compound was not widely detected, implying breakdown in the soil to a more mobile and persistent derivative.

In summation, the following appear to be the key outstanding needs in relation to the assessment of the contamination hazard for British groundwater presented by agricultural pesticide use:

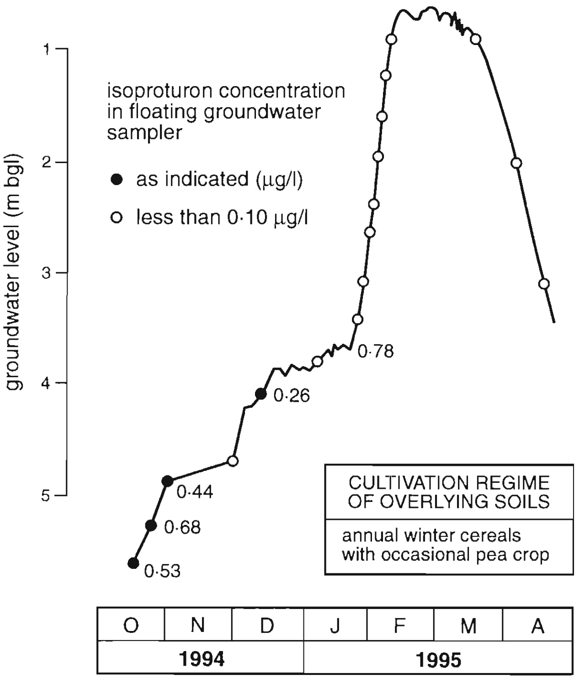


Fig. 6. Variation of isoproturon concentration at the shallow water-table beneath winter cereal cultivation at a Hampshire Chalk Site (Foster & Chilton 1998).

- development of survey methods to identify those areas where the vadose zone of the major aquifers is most prone to frequent preferential flow
- improved understanding of degradation processes and rates for selected pesticide groups in the vadose and saturated zones of the major aquifers
- better knowledge of the occurrence and significance of pesticide metabolites in aquifers
- appraisal of the effect of sampling installations on routine monitoring results for pesticide residues
- assessment of the representativity of laboratory batch and column experiments in relation to field conditions controlling pesticide leaching.

Groundwater Cryptosporidium contamination: and now for something completely different

Nature of concern over Cryptosporidium

Agricultural activity can lead to the generation of microbial and pathogen loading on the soil, and in surface watercourses, by a variety of processes including direct excretion, the discharge of waste from intensive livestock rearing units and the application of sludges and slurries to land as organic fertilizer. In turn, microbes can penetrate into the subsurface, which in more vulnerable aquifers can result in the intermittent detection of fecal coliforms (Escherichia coli) in raw groundwater samples. In the case of the most highly-vulnerable aquifers these faecal coliform peaks can reach in excess of 1000 cfu per 100 ml. Among the pathogens, which may be present in contaminated groundwaters, are some strains of Escherichia coli, certain enteric viruses and the cysts of the protozoa Cryptosporidium parvum and Giardia lamblia. The practice of applying precautionary disinfection to all public groundwater supplies adequately controls most aspects of the associated health risk, but there are in excess of 100 000 private water sources in Britain, which are mostly groundwater-based and receive minimal treatment.

Of greatest concern at present is Cryptosporidium parvum, which infects livestock, various wild mammals and man, with cryptosporidiosis being an endemic disease especially affecting calves and lambs under intensive rearing. In infected animals, the parasite multiplies in the gastrointestinal tract leading to the excretion of very large numbers of oocysts. These oocysts are tiny spore-like structures, which are very resistant to environmental degradation and to standard chlorine disinfection normally used for precautionary treatment at groundwater sources. They thus pose a particularly serious potential hazard. The only effective water-supply treatment is ultrafiltration, involving very high capital and running cost increments when installed at individual groundwater sources. Moreover, current analytical methods for Cryptosporidium parvum are costly and routine demonstration of oocyst viability is not yet reliable.

That contamination of groundwater by Cryptosporidium can occur is now undoubted (Bridgman *et al.* 1995). Of the 12 most recent outbreaks of waterborne cryptosporidiosis in the USA, 4 were traced to contaminated wells (Hancock *et al.* 1997) and 17 of 74 wells in the survey recorded Cryptosporidium parvum (with an average 41 oocysts per 100 l). In Britain a 1997 outbreak north of London with 345 confirmed cases of cryptosporidiosis was traced to a groundwater supply (DWI 1998) and events associated with public-supply groundwater sources have averaged more than one per year (Morris & Foster 2000).

It is suspected that significant under-reporting of contamination incidents may also have occurred in the past, not least due to the absence (until recently) of consistent and reliable industry-wide analytical procedures. However, on recent evidence it can be concluded overall that contamination of raw groundwater supplies by Cryptosporidium parvum is an infrequent, but by no means rare, occurrence in Britain. While in some cases the genotype of Cryptosporidium parvum implicated may indicate an urban (wastewater) source, and in others inadequate sanitary integrity of the source works (or wellhead) itself may be the key factor, groundwater recharge pollution from essentially diffuse agricultural sources is the most significant process.

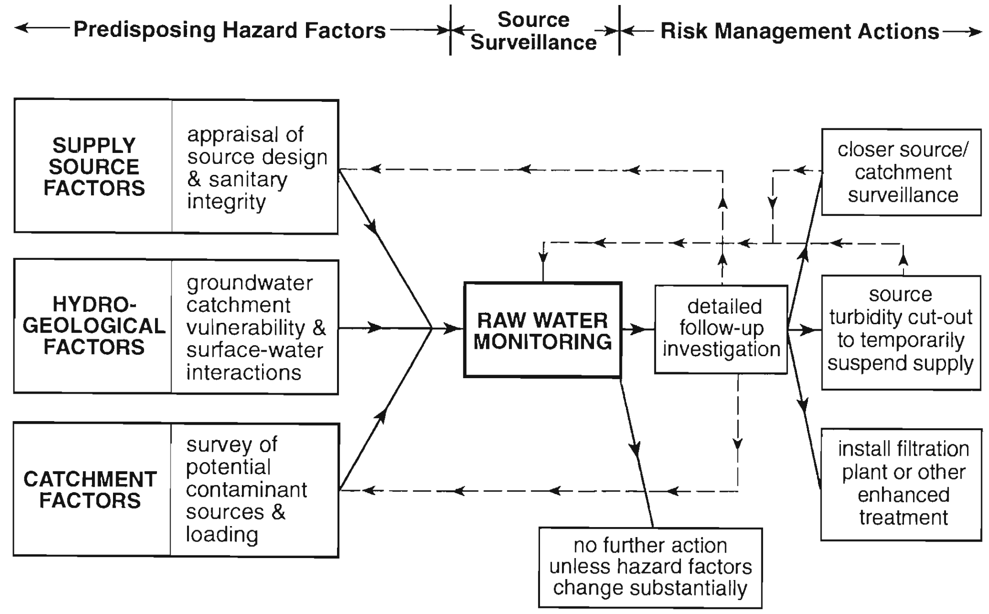


Fig. 7. General scheme for Cryptosporidium contamination hazard assessment and risk management for groundwater sources.

Subsurface transport and persistence of Cryptosporidium oocysts

Cryptosporidium oocysts are 4–6 µm in diameter (compared to 25 µm for Giardia cysts) and larger than the typical 1 µm pore-size of the Chalk aquifer, but they are within the pore-size range of aquifers like the Triassic Sandstone, and are almost certainly smaller than the fissure/micro-fissure apertures which dominate groundwater transmission in the major British aquifers. Thus the capacity to physically detain oocysts in the saturated zone may be limited and the role of preferential flow through the vadose zone especially significant (Fig. 4).

The fate of microorganisms in the subsurface is determined by the interaction of groundwater flow rates and pathogen survival times. At groundwater temperatures of 20°C, a 90% reduction in fecal bacteria can be expected within about 10 days, although a few may persist for 200 days or longer (Lewis *et al.* 1982). Groundwater temperatures in England are typically about 10°C and at this temperature survival of fecal bacteria may be at least twice as long and for Cryptosporidium oocysts (which are more resistant to degradation) probably much longer. Viability of Cryptosporidium parvum in the deep subsurface has not been studied, but oocysts have been reported to survive dormant for months in moist soil (Maudsley *et al.* 1996; Bouchier 1998).

Pollution hazard assessment for groundwater sources

Given the risk of an outbreak from contaminated public groundwater sources, the recent British national expert group (Bouchier 1998) recommended that water companies evaluate the potential hazard of raw (untreated) water contamination in the sources they operate. A flexible (but consistent) methodology for groundwater hazard assessment has been developed, whose conceptual basis contemplates the need to evaluate three sets of predisposing factors (Fig. 7). The 'pathway factor' is pivotal to the hazard assessment, because even if there is significant Cryptosporidium oocyst-loading in the catchment, the hazard will still be low if there is no rapid pathway through the aquifer to the groundwater source. However, the prominence of wells with adits and of spring sources in the list of actual incidents (Morris & Foster 2000) shows that source sanitary integrity also remains a very important consideration.

It might be concluded that all aquifers with an important fissure-flow component, such as the British Chalk, have a high Cryptosporidium pollution hazard. However, there are many Chalk groundwater sources which consistently produce water of very high microbiological quality and very low turbidity. However, there are others, which show intermittent evidence of significant fecal coliform counts and turbidity problems. It is thus important to consider the degree to which karst-like solution features are developed in the catchment area of the groundwater sources (Fig. 8), as a key consideration in hazard assessment. Such features can be detected by direct field mapping, and more effectively in conjunction with 3-D digital photogrammetry to allow landform interpretation. It should be noted that the presence of solution-features anywhere in the source capture area is potentially significant, since even if the average groundwater transit-time to the source exceeds 50 (or 400) days, the presence of more rapid (preferential-flow) pathways is the relevant factor.

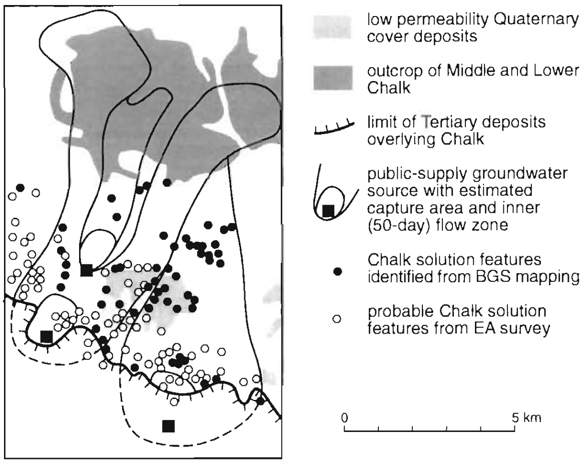


Fig. 8. Mapping of Chalk groundwater source protection areas in part of southern England, showing hazard posed by probable karstic solution features beyond the microbiological protection zone (average 50-day horizontal flow perimeter).

Monitoring indicators of rapid surface water inflow

Monitoring of raw water from groundwater resources is an essential element of hazard assessment (Fig. 7) and risk management. For sources with significant contamination hazard, detailed investigation with automatic samplers and on-line measurement is recommended, using the parameters and approaches in Table 6. Evaluation of the utility of the various possible indicators of Cryptosporidium contamination is constrained, in the case of groundwater sources, by rather limited monitoring data for the parasite itself until very recently.

Although there is no evidence for any consistent correlation between the detection of fecal coliform indicator bacteria and Cryptosporidium oocysts, the regular detection of the former warrants investigation for the latter (Chauret *et al.* 1995). This is particularly the case where there is evidence of rapid groundwater recharge conditions or Cryptosporidium-generating activities within the catchment area. Nevertheless, fecal coliforms can occur in the absence of oocysts and vice versa, because of differences in their temporal distribution in recharge, in their sorption characteristics and the greater persistence of oocysts in the subsurface. There is considerable interest in the feasibility of using the spores of the bacterium Clostridium perfringins as a surrogate for Cryptosporidium parvum in groundwater and also a suggestion that monitoring for fecal streptococci provides a more reliable indicator than fecal coliforms (Edberg *et al.* 1997).

Moreover, several physical and chemical indicators can help establish where the groundwater might be under the rapid influence of surface water. Of these, turbidity is the most convenient and warrants further consideration. The presence of significant turbidity events in groundwater does not necessarily indicate the presence of oocysts, nor does low turbidity confirm their absence. However, sudden unexplained peaks in turbidity should be investigated by use of particle-size analysis and microscopic investigations to distinguish between aquifer material and microorganisms derived from surface watercourses. There is need for research into the mechanisms causing turbidity in groundwater. Such research is currently limited by the restricted deployment of continuous turbidity and particle-size monitoring at groundwater sources.

Logging of micro-temperature and micro-electrical conductivity in boreholes (Table 6) is a well-established technique for identifying the depths and characteristics of major individual groundwater inflows, and chemical parameters (such as chloride, potassium, phosphate, strontium, bromide and boron, as well as stable oxygen and hydrogen isotopes) have been used to determine the proportion of surface water inflow. However, in the upper reaches of streams there may be little difference between surface water and groundwater chemistry, and here hydrogeological transport studies using tracers will be essential to increase confidence in the diagnosis of rapid surface water inflow.

Table 6. *Groundwater source raw-water monitoring data relevant to Cryptosporidium contamination hazard assessment*

Determinand	Type of Sample	Significance
Cryptosporidium oocysts	source water water distribution sediments	direct evidence of active contamination evidence of recent contamination
Escherichia coli	source water (regular detection)	fecal contamination indicator
Clostridium perfringens	source water	possible surrogate for Cryptosporidium
Turbidity counts	source water (temporal changes)	possible rapid influence of inflows of surface water
FT and EC measurements	downhole logging (marked depth variations)	possible major inflows of surface water at shallow depth
Dissolved inorganic species (K, Cl, Sr, PO_4, Br, B, $^2H/^{18}O$, ^3H, CFCs)	marked temporal or downhole variations	indication of recent surface water inflows

Up-date on groundwater nitrate pollution: too much of a good thing!

Nitrate leaching from permeable soils under arable cultivation

Since the initial appearance of the groundwater nitrate problem in lowland England, it has become clear that the leaching of nitrate from essentially-dryland agricultural soils is dependent upon a complex interaction of soil type, cropping regime and rainfall timing. This results in significant uncertainty when estimating average annual rates of leaching loss. Under intensive cereal cultivation in Britain, it is common for 30–60 kgN/ha/a to be leached to groundwater where fertilizer applications are in the range 100–150 kgN/ha/a, and higher leaching losses can occur if the agronomically-optimum application is exceeded. They may also occur beneath potato, oil-seed and some horticultural crops, which receive heavier applications of nitrogen fertilizer (Goulding 1999).

It is also clear that nitrogen fertilizers are subject to complex soil processes and may be taken-up directly by the growing crop, incorporated into the soil nitrogen pool, reduced and lost in volatile form (as ammonium or nitrogen gas) or by soil leaching or in surface run-off (as nitrate or ammonium). Thus, while only a small proportion of the nitrate leached in a given year is derived directly from inorganic fertilizer, the overall rate of nitrogen oxidation and leaching normally relates in a general way to fertilizer application rates. However, some leaching from the soil will occur even if no nitrogen is applied and/or the land is fallow.

Leaching losses from arable agriculture have produced nitrate concentrations in the unconfined groundwater of major British aquifers widely in excess of 50 mgNO$_3$/l, since most of the aquifer recharge areas in eastern and central England are under arable cultivation. From a combination of hydrogeological investigation and agronomic research (using large minimally-disturbed lysimeters with permeable soil profiles), nitrate leaching has been shown to vary considerably with certain facets of arable management practice, such as the timing of fertilizer applications and the soil cultivation methods (Foster 1989). Most nitrate is leached in late autumn when a period of active natural soil nitrification is followed by one of excess rainfall (Fig. 9). Leaching can be reduced (but not eliminated) by ensuring the presence of a growing crop and avoiding fertilizer application during this period.

Effect of pasture management on nitrate leaching

Since pasture is less prone than cultivated land to nitrate leaching, it would appear to offer a useful option for controlling aquifer nitrate pollution. However, organic nitrogen accumulates in pasture land soils and can be oxidized and leached at high rates following ploughing for reseeding (Whitmore *et al.* 1992; Jarvis 1999). Moreover, detailed investigation has demonstrated that leaching rates from pasture on well-drained soils increase abruptly to elevated levels when grassland productivity is intensified by heavy applications of nitrogen fertilizer and high-density grazing (Ryden *et al.* 1984). Where grass is cut and removed for animal fodder, the use of nitrogen is relatively efficient and leaching modest even at fertilizer applications of 250 kgN/ha/a. But if grazed the nitrogen taken-up is excreted and returned to the soil. While some of this is lost to the atmosphere by ammonia volatilization, much of it becomes oxidized and subsequently leached, leading potentially to high rates of loss.

It should be noted that British rates of inorganic fertilizer and organic manure application are only moderate by EC standards, with much higher rates being commonplace in The Netherlands, Belgium and Germany. However, lowland Britain has more widespread occurrence of soils that are susceptible to nutrient leaching and that promote low rates of *in-situ* denitrification, compared to the situation in some of its northern European neighbours.

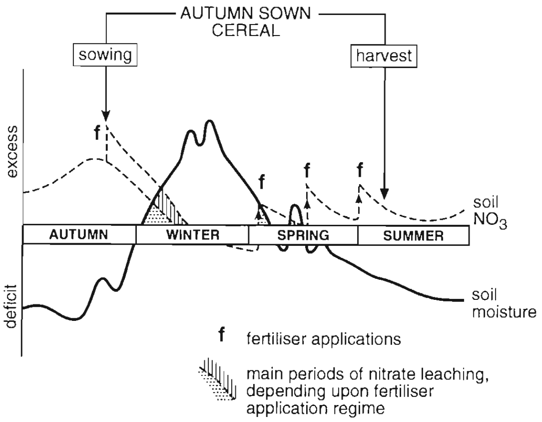


Fig. 9. Generalized regime of winter-sown cereal cultivation showing periods of nitrate leaching (Foster & Chilton 1998).

Subsurface *in-situ* nitrate attenuation capacity

The *in-situ* denitrification capacity of groundwater systems results in removal of nitrate, and thus has been the subject of increasing research in recent years, using a variety of methodological approaches (Table 7). If active on a widespread basis it will have a beneficial effect on groundwater quality and result in less pressure on the agricultural sector to reduce nitrate leaching losses. It is most likely to occur in those aquifers (or parts of) aquifers with a significant natural content of labile

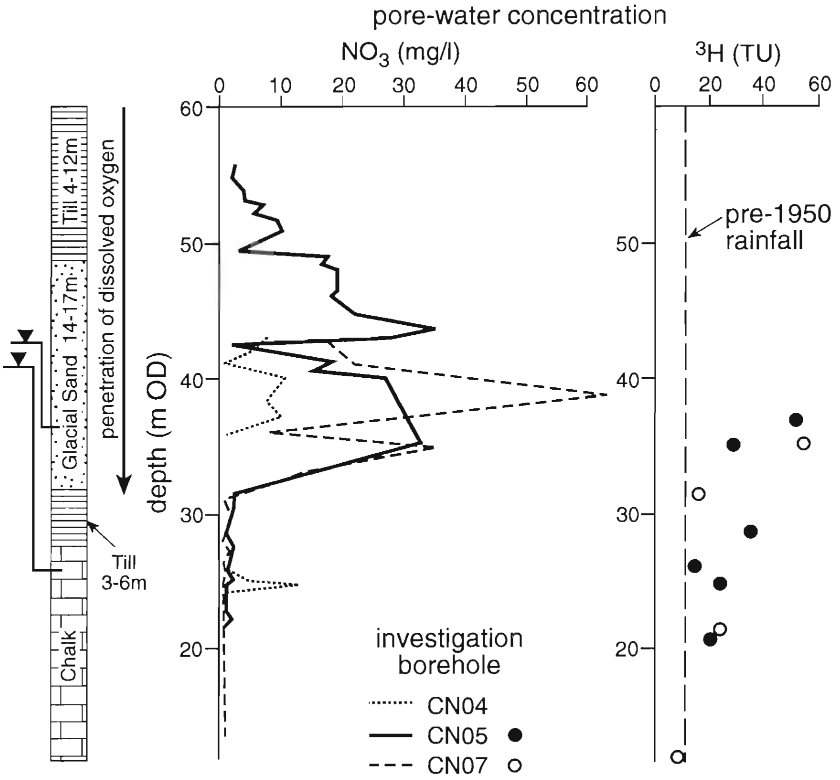


Fig. 10. Pore-water nitrate and tritium profiles through a Glacial Till sequence in Norfolk showing evidence of *in-situ* denitrification (Foster *et al.* 1985).

organic carbon and/or pyritic minerals in the rock matrix (Korom 1992; Starr & Gilham 1993).

Evidence of denitrification comes from the confined (anoxic) parts of some British aquifers (Edmunds *et al.* 1984), and especially through the occurrence of modern tritium-enriched groundwater with negligible nitrate (Lawrence & Foster 1986). The presence of appropriate bacteria was confirmed in core samples and the potential for bacterially-mediated denitrification was demonstrated in laboratory culture experiments, although it is also possible (at least in theory) that the process could be chemically-mediated and accompanying the oxidation of pyrite. Similar transformations are observed in some, but not all, confined parts of other aquifers; there is a particular question, however, as to whether denitrification is active in the confined Triassic Sandstone aquifer, because of very low organic carbon content (both solid and dissolved phase) and the extreme persistence of dissolved oxygen in groundwater.

There is further evidence of *in-situ* denitrification in situations where the Chalk aquifer is semi-confined by overlying Glacial Till deposits (Fig. 10) (Foster *et al.* 1985). Here the evidence is both from the downward penetration by leakage of tritium-enriched recharge with zero nitrate content and the presence of appropriate viable bacteria.

It should, however, be noted that the proportion of the total national groundwater resource involved in flow

Table 7. *Summary of methodological approaches for the evaluation of denitrification in groundwater systems*

Methodology	Limitations
Mass Balance Studies	
*repeated vadose zone profiling	natural spatial variability of individual profiles
*periodic 3-D surveys of water chemistry	purpose-drilled network of monitoring boreholes needed
Distribution of Key Constituents (NO_3, NO_2, NH_4, Bacteriological Counts, DO, DOC)	indirect (often inconclusive) evidence
Monitoring of Free/Dissolved Gases (N_2O, N_2/Ar, CO_2/CH_4)	rarely conclusive, except in 'closed systems'
Key Isotopic Indicators (^{15}N‰; ^{18}O‰ in NO_3 and H_2O)	also varies with genesis
Evidence of Soil/Rock Fabric (atrophic denitrification with FeS_2 oxidation)	oxidation by other processes but needs catalyst

into deep confined aquifer systems or in recharge through semi-confining beds is relatively small. Recent interest, therefore, has focused on the potential for denitrification in the vadose zone of unconfined aquifers. This subject has been intensively researched at a range of research sites—especially Bridget's Farm on the Upper Chalk in Hampshire and at Gleadthorpe on the Triassic Sandstone in Nottinghamshire (Kinniburgh *et al.* 1999). Although some potential for denitrification was observed (especially at the base of the soil zone and in the zone of seasonal water-table fluctuation), there was little conclusive evidence of denitrification at significant rates. The following was concluded:

- fairly high concentrations of nitrate were present throughout the vadose zone, but despite the presence of potentially-denitrifying bacteria (10–1000 cfu/ml and higher in subsoil) and significant levels of dissolved organic carbon (2–10 mg/l) there were only occasional indications of the generation of nitrite or ammonium
- monitoring of both dissolved and free gases (including O_2, CO_2, CH_4, N_2O) did not provide evidence of significant denitrification; O_2 levels were fairly constant at 18–21% and although dissolved N_2O was somewhat greater than expected in the vadose zone this could have been generated at the water-table equally by nitrification as by denitrification
- the N_2/Ar ratio in the gas phase showed little increase above open atmosphere levels, and assuming a closed system could only explain denitrification at rates of less than 4 kgN/ha/a.

Status of national aquifer monitoring: a very unsatisfactory affair!

Current monitoring deficiencies

Although there has been continuing investment in understanding the processes controlling the leaching of agriculturally-derived contaminants to groundwater, the situation in terms of monitoring quality trends in groundwater recharge to the major aquifers is far from satisfactory. No equivalent exists in Britain to the purpose-drilled groundwater monitoring networks in some of our northern European neighbours, such as The Netherlands, Denmark and Germany. Indeed, it is not an exaggeration to say that:

- the existing groundwater quality monitoring network is not adequate to provide information on the detection and control of diffuse pollution from agriculture
- the current level of monitoring installations in the main aquifer recharge areas are not sufficiently refined even to indicate whether the trend for nitrate and some important pesticides in present recharge is actually rising or falling with time!

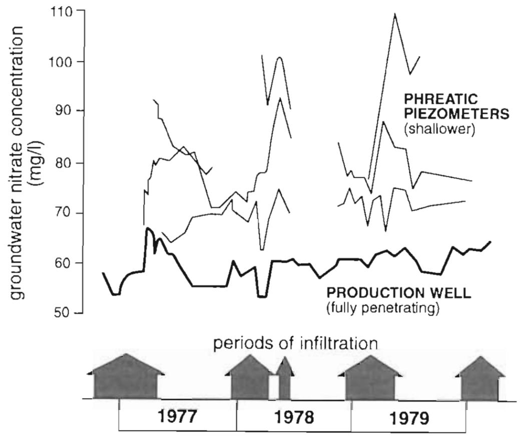


Fig. 11. Detection of rising trend in groundwater nitrate concentration from diffuse agricultural pollution through monitoring with shallow phreatic piezometers in the Norfolk Chalk aquifer.

There are two reasons for this unsatisfactory state of affairs:

- over-dependence on quality monitoring in water-company production wells
- inadequate coverage of purpose-drilled aquifer recharge monitoring boreholes.

Dependence on samples from deep production wells must be regarded as 'post-mortem monitoring' as far as most aquifer systems are concerned. This is because such boreholes invariably pump a 'cocktail' of groundwater of very different origin in terms of recharge date and residence time. This, together with the lack of clearly specified protocols for raw-water sampling with respect to pumping regime, means that such monitoring provides highly insensitive datasets with which to interpret aquifer recharge quality. There are a few exceptions (almost all in the 'water-supply only companies') where raw-water samples have been taken and analysed on a regular and consistent basis for many decades. However, in general it can be clearly shown (Fig. 11) that shallow monitoring boreholes are far more effective for the early identification of trends in groundwater recharge quality in situations where land-use pressures are increasing the concentration of anthropogenic contaminants. The coverage of such monitoring boreholes is, however, wholly inadequate and only locally do such installations exist.

Up-grading the national monitoring network

A minimal, cost-effective, up-grade scheme for the national groundwater quality monitoring network,

Table 8. *Hierarchy of British control measures relevant to diffuse groundwater pollution by agricultural practices*

Legal Provisions	Primary	EC Directives
	Secondary	Environment Act – Groundwater Regulations EC Nitrate Vulnerable Zones WR Act – Pollution Offences
Regulatory Policies	EA Policy and Practice for Protection of Groundwater	
	MAFF Registration Requirement for Agrochemicals	
Operational Guidance	MAFF Guide on Best Agricultural Practice for Water Protection	
	EA Definition of Groundwater Source Protection Areas	
	MAFF Farm Action Plans for Nutrient Management and Chemical Handling MAFF/EA Farm Environmental Management Demonstration Projects	

including improvement of sampling protocols for production wells and the installation of a national network of shallow monitoring boreholes was produced in 1994 by the British Geological Survey (BGS) in cooperation with the (then) National Rivers Authority. However, investment in such a network has not received the necessary priority from the Environment Agency (EA) or the Department of Environment, Transport & Regions (DETR) and no significant progress has as yet been made on this front. The low priority would appear to reflect a lack of consciousness of the need and value of reliable aquifer recharge quality data as part of the national effort on 'state of environment monitoring' and also the fact that there is currently no regulation requiring compliance in this respect. One significant exception has been recent investment by the EA (jointly with the BGS) to evaluate baseline groundwater chemistry and this will serve as a useful foundation for strengthening national groundwater quality monitoring.

The situation is changing as a result of the implementation of the EU Nitrate Framework Directive and will need to change further with the introduction of the EC Water Framework Directive, which clearly specifies an obligation on member states to monitor trends in the quality of aquifer recharge.

It is strongly recommended that the DETR should mobilize appropriate funding for up-grading this network and give responsibility to the BGS for long-term maintenance of such datasets. Obviously the BGS would need to work in close collaboration with the EA, but the role of the BGS should be to ensure that adequate long-term datasets are collected in the interests of all groundwater users, and not merely the regulatory need of the EA itself. An arrangement of this type currently works well for groundwater level data, where the EA and the BGS work in close cooperation to ensure adequate coverage and consistency of long-term monitoring, together with data accessibility to a wide range of stakeholders.

Control of diffuse agricultural pollution: something of a policy quagmire

Obstacles and difficulties for pollution control

Farming is an inherently 'leaky activity', particularly when practised on permeable sub-soils generating significant preferential vadose-zone flow. While the corresponding regulatory agencies have a range of legal provisions, together with supporting regulations and guidance (Table 8), as of yet there has been only limited success in controlling the problem of diffuse groundwater pollution from agriculture practices.

Problems have been encountered for a variety of reasons including:

- current legislation is focused more on the control of point (than diffuse) source pollution
- inspection and verification present substantial cost and technical difficulty
- the EA is not directly empowered as regards microbiological contaminants
- the EA would prefer to act on a risk-based approach, which implies control applied to selected areas of land (those most vulnerable to pollution and located in source protection areas), however, while in theory they have legal provision to act in this way, in practise this has proved extremely difficult without payment of compensation to farmers for increased cost and/or supposed loss of agricultural production
- the EA is encouraged to act through the polluter-pays-principle, but this in practice is impractical and tardy in relation to diffuse agricultural pollution.

Groundwater contamination by pesticides and microbial pathogens presents rather different problems to that by nitrates, since they are generally much less persistent and more related to the incidence of rapid preferential flow. Nevertheless, pollution control also poses formidable challenges.

At the macro-policy level it has to be recognized that the current EU Drinking Water Guidelines for both nitrates and individual pesticide compounds are highly idealized and little justified on health and economic grounds. They act as a disincentive (rather than an incentive) for groundwater protection, because it is widely perceived that protection to such levels is not possible without the abandonment of modern agriculture, and thus treatment of groundwater supplies is the only reliable solution. As regards nitrate, guidelines on best agricultural practice cannot be fully effective in reducing the average concentration in infiltration from cultivated land to below 50 $mgNO_3/l$, and there will be certain situations where the average rate of leaching will lead to concentrations 50–100% above this level.

Ironically at the same time neither the EA (and for that matter current provisions in the EC Water Framework Directive) have (or anticipate) powers for the control of the discharge of microbiological contaminants, which complicates the problem of reducing the discharge of Cryptosporidium oocysts to the environment. As regards pesticides, it appears on the current state of knowledge that the only way to guarantee that groundwater recharge contains less than 0.1 μg/l would be to prohibit the use of the most mobile compounds in areas of high aquifer vulnerability with frequent preferential vadose-zone flow.

At the heart of the long-term problem also is the fact that the EU Common Agricultural Policy (CAP) has over many years subsidized certain types of agricultural production (notably cereal, oil seed, dairy and meat products) in areas far from ideally-suited environmentally for such production. The CAP has also not encouraged governments to undertake field-scale trials before introducing new subsidized cultivation regimes, to establish what impact they might have in terms of the leaching of nitrates and pesticides to groundwater. Diffuse pollution control has also not been aided by the British public's (and farmers') general attitude on food, which until recently could be summarized as 'price before quality'.

Progress achieved since mid-1980s

Some significant progress has, however, been made in a number of senses (Chilton & Foster 1991). First, it has been clearly recognized by the agricultural sector that fertilization regimes needed to be controlled in the interests of reducing leaching and protecting groundwater quality. Following pressure from the water sector in general (and groundwater scientists in particular) the Ministry of Agriculture Fisheries & Food (MAFF) has mounted an unequivocal campaign to educate farmers in respect of nitrate leaching by:

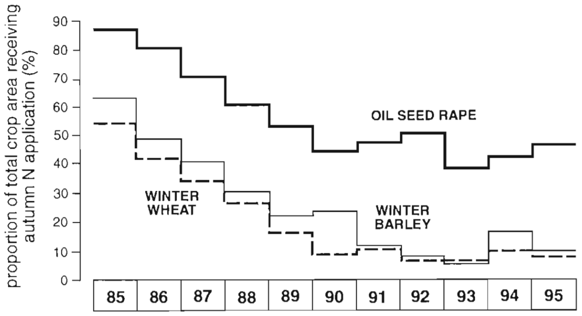


Fig. 12. Reduction in autumn nitrogen applications to major arable crops in England and Wales during 1985–95 (Dampney *et al.* 1999).

- avoiding the application of nitrogen fertilizers in autumn, when there is invariably an excess of nitrate in the soil in relation to most plant requirements
- applying nitrogen top dressings only during active plant growth, taking account of seasonal factors such as summer drought on grassland
- promoting the calculation of nitrogen requirements for individual fields in relation to crop type, expected yield and soil nitrogen supply
- encouraging the sowing of autumn crops (in preference to spring-sown crops), and as soon as practicable after harvest, to ensure active plant growth during the autumn period of maximum nitrate soil availability
- considering the use of cover crops to reduce leaching during the winter months (this is mandatory in some EC countries)
- encouraging minimal soil disturbance, by reducing soil ploughing and using direct drilling where possible, and failing this incorporating straw to conserve soil organic matter and nitrogen retention capacity.

Such measures have led to less use of nitrogen fertilizers in Britain (and in Europe) and it is estimated that a 30% reduction has been achieved since 1985. They have also resulted in a marked reduction in the area of fallow land in autumn. The effectiveness of advice to avoid application of nitrogen fertilizers during September–February can be judged from recent data (Fig. 12).

Some progress has also been made on the control of agricultural pesticide use in the interests of groundwater protection. It is now normal practice for the MAFF to insist upon laboratory and field trials in relation to compound mobility before product approval, and also to encourage the development and substitution of less mobile compounds wherever possible. However, little progress has been made on controlling the application of more troublesome compounds within groundwater protection areas. Some water companies have considered reaching private agreements with farmers cultivating their groundwater source protection areas, for example to pay compensation for not using compounds such as simazine for weed control. However, these arrangements have proved difficult to pursue in practice without the active involvement of the EA and the MAFF.

Agriculture and groundwater in the next 25 years: opportunities, threats and unknowns

Changing face of British agriculture

There is little doubt that agricultural practices and rural land-use in lowland Britain are likely to continue to experience major changes in the next 25 years. How will these influence groundwater?

Some of the key unknowns include the rate and spread of organic and ecosystem farming. It is to be hoped that market forces and changes in the CAP will provide incentives for agricultural production regimes which are less dependent on high applications of agrochemicals. However, while this type of farming regime will greatly reduce the risk of groundwater pollution by pesticide residues and pathogenic micro-organisms, it will not necessarily eliminate significant nitrate leaching, since this will depend on the detail of cultivation processes and crop rotations selected.

Will the current trend of taking land out of agricultural production for conservation and recreation use continue? This is also uncertain; but current planning procedures do not allow land to be targeted in areas of high groundwater vulnerability and/or source protection areas, where the greatest benefits in terms of improved groundwater recharge quality would be felt. A further relevant issue will be the national response to predicted future world food deficiencies, which might lead to increased agricultural production for export purposes.

Another major uncertainty is the potential impact of accelerated climate change. This could modify soil leaching regimes, lead to changes in crop type and to the extent that supplementary irrigation that is practised. These changes could possibly increase the risk of serious leaching of agrochemicals to groundwater, unless appropriate controls are enforced.

Certain issues associated with the potential impact of livestock rearing practices are already with us, although the threat which they pose to groundwater quality is not yet clearly established. These include:

- the nutrient loading arising from the intensive rearing of pigs (and to lesser degree other livestock) in open-air conditions in aquifer recharge areas
- the handling and disposal of livestock effluents, which may contain high concentrations of Cryptosporidium oocysts, Escherichia coli-0157 and other microbial contaminants
- the possible infiltration of elevated concentrations of hormonal and antibiotic compounds intermittently observed in effluents generated by livestock rearing activities.

These are all subjects, which require field research.

Another problem that may already be occurring is transport via the groundwater route of agriculturally-derived phosphate. Little monitoring or research on soluble (or particulate) phosphate in British aquifers has been carried out because concentrations nowhere approach the EC drinking water guideline (2.2 mgP/l). However, much lower concentrations may be environmentally-significant due to triggering of algal blooms and eutrophication in aquifer-fed surface water bodies.

Potential benefits of technological innovation

It is also likely that agriculture will experience an accelerated rate of technological innovation in the coming decades, with the introduction of 'new chips, new genes and new molecules'. This will result both in significant opportunities, but also significant threats, where diffuse pollution of groundwater is concerned. The most significant changes are likely to be:

- the increased availability of low-cost sensors (backed by computers) which could enable micro-management of soil moisture and agrochemicals at farm and field level, with the opportunity of much more control over soil leaching losses
- the possibility of designing new pesticide molecules which are far less mobile in soil solution and thus less prone to leaching to groundwater
- the potential widespread introduction of genetically-modified crops, which could be far more efficient in terms of nutrient uptake: however concern has to be expressed because these may require much higher applications of specific pesticides and have a potential effect on soil bacteria and soil contaminant attenuation.

At the level of hydrogeological research, it is hoped that uncertainty in the quantification of attenuation capacity of the vadose and saturated zones for specific agrochemicals will be significantly reduced and that this attenuation capacity will be used intelligently in the interests of protecting groundwater quality. At the environmental management level, it is hoped that selective risk-based land-use controls will become more acceptable to the farming community and that special taxes on fertilizers and pesticides could be raised to provide funds for compensation of those affected.

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