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Grassed buffer strips for the control of nitrate leaching to surface waters in headwater catchments

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Abstract

The use of riparian buffer strips is a possible strategy for controlling diffuse nitrate pollution of surface water in agricultural catchments. Data collected from paired buffered and unbuffered headwater catchments at three sites with conditions representative of much of the agricultural land in England and Wales, showed that grassed buffers did not substantially reduce nitrate–nitrogen concentrations entering the streams. Median nitrate–nitrogen levels observed in buffered catchments ranged from 7.6 to 18.8 mgN l^{-1}, but peaked at up to 46.1 mgN l^{-1}. The existence of preferential bypass flow paths during the winter flow events limited the effectiveness of nitrate removing processes within the strips. The findings suggest that grassed riparian buffer strips may not be effective in controlling diffuse nitrate pollution unless the hydrology of the strip allows a suitable environment for denitrification and/or plant uptake. Grassed buffer strips should be carefully targeted or, alternatively, engineered to ensure adequate residence time of solutes within the strip. © 1999 Published by Elsevier Science B.V. All rights reserved.

Keywords: Grassed buffer strip; Nitrate removal; Hydrology; Headwater catchments

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1. Introduction

Concentrations of nitrate in surface waters in the UK have increased steadily since the 1960s (Addiscott et al., 1991), mainly as a result of agricultural intensification. A major factor has been a growth in the use of inorganic nitrogen fertiliser, which has enlarged the overall organic nitrogen pool in agricultural soils. Autumn tillage after harvesting, when conditions are warm and moist, stimulates soil microbial activity, resulting in the mineralisation of this organic nitrogen into nitrate. Nitrate is highly soluble and, in the absence of crop cover, is leached from agricultural land as the soil re-wets. As a result, nitrate levels in surface waters in the UK often peak during the first winter storm flows in November and December.

Concerns about the possible effects of nitrate in drinking water on public health have led to a limit of 11.3 mgN l^{-1} of nitrate–nitrogen in potable water being set by the EC Drinking Water Directive. This necessitates increased expenditure on water treatment by public water suppliers in areas where surface and groundwater nitrate levels are high. Nitrate pollution also poses a threat to the natural environment. Nitrate is the limiting nutrient for eutrophication in seawater, and is also responsible, to a lesser extent, for freshwater eutrophication. It is therefore important to investigate methods of reducing the levels of nitrate entering surface waters.

While nutrients originating from point sources may be relatively easy to target and control, diffuse pollution is difficult to prevent. Riparian buffer strips, permanently vegetated with trees, scrub or grass, have the potential to improve water quality by reducing soluble nutrient concentrations, sediment loads, and sediment-bound pollutants entering watercourses in run-off from agricultural land (Haycock and Pinay, 1993; Dillaha and Inamdar, 1997; Gril et al., 1997; Uusi-Kämppä et al., 1997). For example, Haycock and Burt (1993) found that nitrate concentrations were reduced by 84% in shallow groundwater in a transect across a grass buffer strip in the Cotswolds. Nitrate may be removed from shallow groundwater flowing laterally through the root zones of buffer strips by vegetative assimilation and denitrification. Perennial grasses may assimilate up to 400 kg N ha^{-1} yr^{-1} (Addiscott et al., 1991). Denitrification occurs when soil microorganisms utilise nitrate as an electron acceptor for respiration under anaerobic conditions, resulting in the release of gaseous nitrogen or nitrous oxide. Stepanauskas et al. (1996) report that the potential for denitrification in soils is large. Waterlogged soils kept at 25°C in the laboratory and supplied with easily decomposable organic carbon may utilise nitrate at a rate of 30 kg N ha^{-1} day^{-1}. However, in situ, the rate is more likely to be less than 1 kg N ha^{-1} day^{-1} (Smith et al., 1994; Maltby et al., 1996), and will depend on factors such as temperature, soil redox potential and the availability of organic carbon.

Investigations by Haycock and Burt (1993) on the role of grassed riparian strips have reported large reductions in nitrate concentrations in water seeping to a stream in a small arable catchment buffered on both banks by grass. In order for buffer strips to be effective sinks for nitrate–nitrogen, suitable conditions must prevail in the strip so that denitrification and/or plant uptake can occur. An energy source for anaerobic bacteria must be present and a suitable redox status must be achieved for the buffer strip to have a chance of removing nitrate from the flowing water by denitrification. The presence of plants, e.g. grasses or trees, within the strip may provide the energy source and long percolation times leading to saturation may ensure anaerobic conditions so, at times of active growth, there is also the possibility of nitrogen uptake and immobilisation by plants within the strip. In addition, long retention times for water within the strip provide greater opportunities for biochemical nitrate removal. However, Hill (1996) points out that riparian zones have less effect on nitrate removal in hydrogeologic settings where groundwater has little interaction with vegetation and sediments.

Haycock and Pinay (1993) suggest that buffer strips encourage slow percolation and long retention times, essential conditions for effective nitrate removal by both assimilation and denitrification. Parsons et al. (1994) point out that increased hydraulic roughness across the strip can decrease overland flow velocities, while permanent vegetation may improve soil structure, encouraging slow matrix flows rather than macropore flow, and increased infiltration, reducing run-off and increasing saturated groundwater flow. Soil left untilled may additionally have fewer tillage induced fissures, which can act as preferential flow paths. The denitrifying potential of buffer strips is also possibly enhanced by increased organic carbon levels (Jenkinson, 1988), resulting from the establishment of grasses in the absence of disruptive tillage.

Buffer strips have attracted considerable attention from environmentalists as a potential pollution control measure, and their installation in designated areas in the UK is currently being encouraged with financial incentives (Ministry of Agriculture, Fisheries and Food, 1996). However, although the effectiveness of nitrate removing processes in the buffer strip is likely to be influenced by the hydrology of the catchment and the buffer strip itself, few catchment-level studies of the effect of buffer strips on surface water nitrate levels have been conducted in the UK.

This paper reports the findings of a study conducted on buffer strips installed in three headwater catchments to investigate whether they could reduce nitrate levels in surface waters in headwater catchments. This formed part of a larger study carried out between 1993 and 1996, which investigated the effectiveness of grassed buffer strips in controlling a range of diffuse pollutants, including phosphate and sediment.

2. Methods

2.1. Study sites

Paired first- or second-order headwater catchments of similar size were selected in Bedfordshire, Oxfordshire and Shropshire (Fig. 1). These sites represented a range of typical soils and climate found in English and Welsh conditions. All of the study areas were under arable cropping with spring fertiliser application, and the watercourses were known to have high nitrate levels. Air photograph interpretation and digital mapping techniques were used to estimate the area of each catchment. In one of each pair, grass buffer strips ranging from 5 to 50 m in width were established along both sides of the stream. The second catchment of each pair was left unbuffered. The effectiveness of the strips was evaluated by comparing stream water flows and water quality in each pair over an 18-month period. The grassed buffer strips were established in a conventionally tilled and prepared seedbed on soil that was previously part of the arable cereal field. The strips were sown with a general seed mix consisting mainly of *Lolium perene* L. (perennial ryegrass) and *Festuca rubra* L. (red fescue) in September and October 1994. Monitoring of the stream water quality was undertaken in the winters of 1994–95 and 1995–96.

2.1.1. Bedfordshire

The Bedfordshire catchments lie between 58 and 75 m above sea level, with gently sloping land. Both are underlain by up to 60 m of chalky boulder clay. The nearest aquifer lies around sea level, and is confined by the overlying clay. Soils are clayey, calcareous pelosols of the Hanslope series and, to a lesser extent, clayey

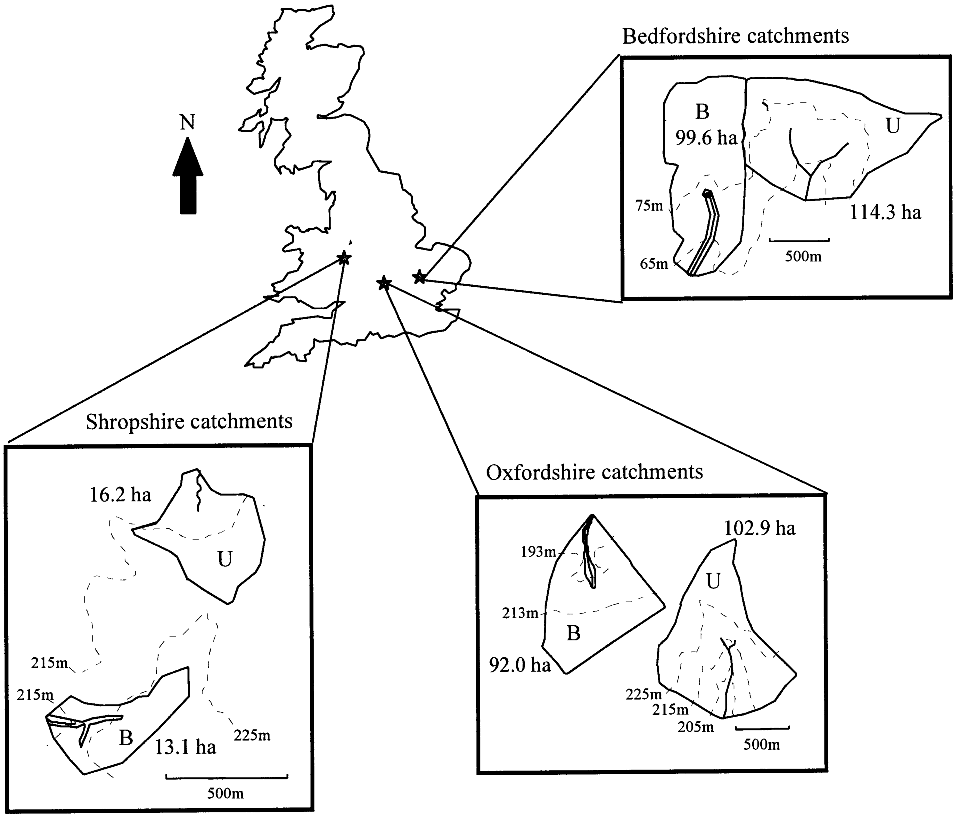

Fig. 1. Site locations and headwater catchments for the three areas investigated. Shaded areas indicate the buffered streams.

pelostagnogley soils of the Ragdale series. Data in the Soil Survey and Land Research Centre's Land Information System (LandIS) indicates that these soils are hydrologically similar to those covering approximately 35% of England and Wales. The Hanslope soils have progressively restricted permeability below about 0.5 m, resulting in seasonal waterlogging. Deep shrinkage cracks appear during the summer, the soil only slowly swelling in late autumn and winter as re-wetting occurs. The Ragdale soils have only slowly permeable subsoils, and remain at or near saturation for substantially longer than the Hanslope series. Soil moisture deficits produce some cracking in summer. This is one of the driest parts of the country. Mean annual rainfall is 550 mm, of which 125 mm is lost to drainage and is said to be hydrologically effective (Smith, 1976). Field capacity conditions persist for an average of 107 days, and the mean accumulated maximum potential soil moisture deficit is approximately 200 mm (Hodge et al., 1984).

2.1.2. Oxfordshire

Located in the Cotswolds, these catchments are between 167 and 235 m in altitude; the unbuffered catchment being slightly higher than the buffered. The highest ground of both catchments is almost flat, with slopes gradually steepening into the valleys, to a maximum of 8–10°. Limestone aquifers are underlain at varying depths by impermeable silty shales of the Upper Lias age, which have small but significant outcrops in both catchments. Soils over the limestone are porous, freely draining stony and clay loams, sandy silt loams or sandy loams. About half are shallow calcareous rendzinas of the Elmton and Marcham series. Decalcified deeper brown earths of the Dinorben and Waltham series are also widespread. The hydrological circumstances of these freely draining rendzinas and brown soils over limestone are representative of soils covering about 23% of England and Wales. On the slopes of the Lias outcrop, fine, loamy, stagnogleyic argillic brown earths of the Bursledon series are most extensive, with some areas of fine, silty, stagnogley soils of the Stanway series. The subsoils of both of these are subject to waterlogging, with periodic saturation of the upper horizons in the Stanway soils. Average annual rainfall is 750 mm, of which 286 mm is hydrologically effective (Smith, 1976). The field capacity season is 160 days long, and accumulated potential soil moisture deficit averages 150 mm (Jarvis et al., 1984). Denitrification potential of the soil at a depth of 0.3 m was measured only on the Oxfordshire buffered site and found to be typically 21 kgN ha^{-1} day^{-1} (Matchett, 1998).

2.1.3. Shropshire

The Shropshire catchments are between 200 and 231 m above sea level. The highest ground is flat or very gently sloping, with the steepest slopes found on the valley sides at the lower ends of the catchments. Both of the watercourses apparently originated as agricultural ditches, and have incised gullies which, in places, exceed 2 m in depth. Both catchments are on Devonian Marls, micaceous silty shales and mudstones. Limestone bands, and some thin sandstones and shales, are interbedded with the marls. No significant aquifers or groundwater are present. The most common soils are silty, stagnogleyic argillic brown earths of the Middleton series, with clay content increasing below depths of 0.4 m and turning to marl or mudstone at about 1 m. Hydrologically similar soils to these Middleton soils occupy about 35% of England and Wales. Soil properties vary considerably. About a third of observations have strong gleying, consistent with surface water gley soils of the Netchwood series. Clayey and loamy textured horizons and profiles, and a few shallow, stony, loamy soils formed from weathered sandstone bands, are also found. The Netchwood soils remain waterlogged for much of the winter, whereas profiles over sandstones are freely draining. This area has the wettest climate of the three sites. Mean annual rainfall is 800 mm, with 333 mm falling as hydrologically effective winter rain (Smith, 1976). Field capacity conditions persist, on average, for 175 days, with the mean accumulated potential soil moisture deficit reaching 125 mm (Ragg et al., 1984).

2.2. Monitoring, sampling and analysis procedures

At each site, 90° V-notch weirs, designed to British Standard 3680 for a 100-year flood (based on the Flood Studies Report, Natural Environment Research Council, 1975), were installed. Water levels were measured hourly by a float and weight set up in a stilling well, attached to a 10-turn potentiometer which was connected to a Campbell CR 10 data logger. The data logger was programmed to control a Rock and Taylor 48 Sample liquid sampler, which collected a water sample every 24 h, unless the water level exceeded a pre-set height (for example, during storm flow events) when the logger instructed the sampler to collect samples every 4 h. The water sampler was reset at regular intervals. Empty replacement bottles contained a mercuric chloride preservative to prevent microbial activity influencing nitrate concentrations. At the buffered catchments, a tipping bucket rain gauge was also attached to the logger, sending a pulse to the logger for each tip of the bucket. The logger recorded the number of tips every 5 min, providing detailed information on each rainfall event. Two temperature probes took hourly measurements of the buffer strip temperature at 100 mm depth and stream temperature; this data was again recorded in the logger. Data was regularly downloaded from the logger, and the storm flow threshold was checked and updated as necessary.

Water analysis was undertaken after first filtering the samples through a 0.45 mm pre-washed filter paper. Nitrate–nitrogen was then determined using a Technicon continuous flow auto-analyser. In this method, nitrate is first reduced to nitrite with hydrazine sulphate under alkaline conditions using copper ions as a catalyst. The nitrite ions then react with sulphanilamide under acidic conditions to form a diazo compound. The compound then couples with *N*-1-napthylene-diamine dihydrochloride to form a reddish purple azo-dye that is measured at 520 nm (Technicon Industrial Systems, New York). Quality control was maintained by inserting a standard every 10th sample. In addition, independent quality control checks were performed at 3-monthly intervals.

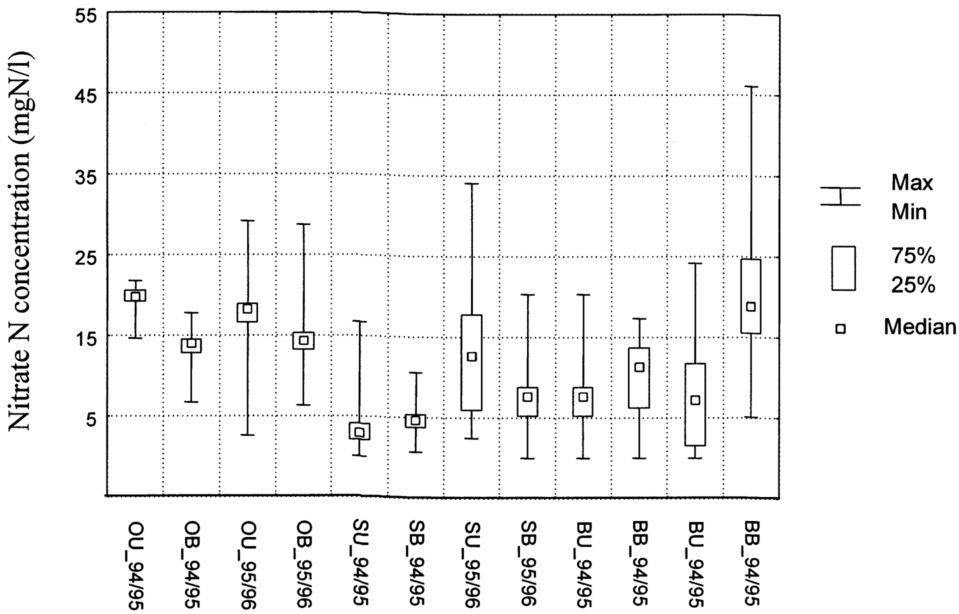

Fig. 2. Summary of nitrate concentrations (winters of 1994–95 and 1995–96): OU, Oxfordshire unbuffered; OB, Oxfordshire buffered; BU, Bedfordshire unbuffered; BB, Bedfordshire buffered; SU, Shropshire unbuffered; SB, Shropshire buffered.

3. Results and discussion

3.1. Water chemistry

Fig. 2 summarises the nitrate–nitrogen concentration data for the 1994–95 and 1995–96 winter seasons, showing the median, maximum and minimum values, and the quartile range to indicate the spread of data. The nitrate–nitrogen concentrations measured at the stream outlets are generally higher than the typical values of 5–10 mgN l^{-1} for lowland rivers in the UK (Betton et al., 1991). Levels measured at all sites considerably exceeded the EC Drinking Water Directive limit of 11.3 mgN l^{-1} at some point during the measuring period. Maximum concentrations exceeded 20 mgN l^{-1} in all catchments, and were as high as 46 mgN l^{-1} in the case of the Bedfordshire buffered catchment. Four of the sites also had median nitrate–nitrogen levels above the permitted threshold, including the buffered Bedfordshire and Oxfordshire catchments.

During the 1995–96 season, both the Shropshire and Oxfordshire buffered catchments had lower median and maximum concentrations than their corresponding unbuffered catchments, initially suggesting that the buffer strips have some effect on the nitrate–nitrogen concentrations leaving the catchments. The opposite situation was found in the Bedfordshire catchments during the same period. During the 1994–95 season, the buffer strips were still developing and would have been unlikely to have had any effect upon pollutant concentrations. However, the median and maximum nitrate–nitrogen concentrations were again found to be lower in the Oxfordshire buffered catchment than in the unbuffered catchment. The Bedfordshire site also follows the same trend during 1994–95 as in 1995–96. The differences in nitrate–nitrogen concentrations from buffered and unbuffered catchments observed during the following season could therefore be attributed to differences in catchment characteristics rather than to the action of buffer strips, making comparisons between paired catchments difficult to interpret. For example, differences in catchments' soil hydrology and drainage may affect the degree of autumn nitrogen mineralisation. Comparisons between catchments in the different geographical areas are unwise, as their hydrological and physical characteristics differ.

There is also evidence of intra-catchment variation between years. The 1994–95 data from the Shropshire catchments shows the opposite trend in nitrate–nitrogen levels to the following year, with higher median nitrate–nitrogen concentrations in the buffered catchment than in the unbuffered catchment. In both the Oxfordshire and Shropshire unbuffered catchments, nitrate–nitrogen concentrations also fluctuate considerably between the 2 years. Such annual fluctuations are to be expected, due to natural variations in seasonal rainfall and temperature regimes, which may affect both mineralisation rates and autumn ploughing and planting dates, and thus the magnitude of winter nitrate leaching. In addition, microbial nitrate production is sensitive to the amount of nitrogen fertiliser applied to the crop in the previous season. Inconsistencies in such crop management practices are difficult to avoid in on-farm trials, which depend on farmers' co-operation, making the imposition of strict controls difficult. Between-year comparisons of data from the same catchment are therefore also unreliable.

The biased and irregular nature of the data, and the short duration of the data record preclude simple statistical tests of difference and make it difficult to draw firm conclusions about the effect of the buffer strips on nitrate–nitrogen concentrations leaving the catchments. However, no substantial improvements in stream water quality are evident due to the presence of buffer strips.

The most likely explanation for the apparently limited effectiveness of the buffer strips in controlling nitrate pollution in these catchments are the short retention times, which are discussed in the following section.

3.2. Hydrology

The literature suggests that the imposition of buffer strips within a catchment may have a retarding effect on the hydrology of the catchment, so providing a long retention time in the buffer strip (Haycock and Pinay, 1993) due to the cessation of autumn tillage and long-term soil structure changes. Long retention times may be needed to ensure that biochemical changes and plant uptake of nitrate can occur. In addition, long retention times lead to wetter conditions, favouring denitrification.

All of the streams studied were characterised by negligible summer flows during the very dry summer of 1995. Significant flows only occurred during periods of excess winter rainfall. Hydrographs for all of the catchments showed a rapid response to rainfall events. As an example, Fig. 3a shows the winter 1995–96 hydrographs for the Bedfordshire catchments. In this case, field capacity conditions leading to stream flow did not occur until late December 1995. This winter was unusually dry, and only four flow events are seen to occur, although the rainfall

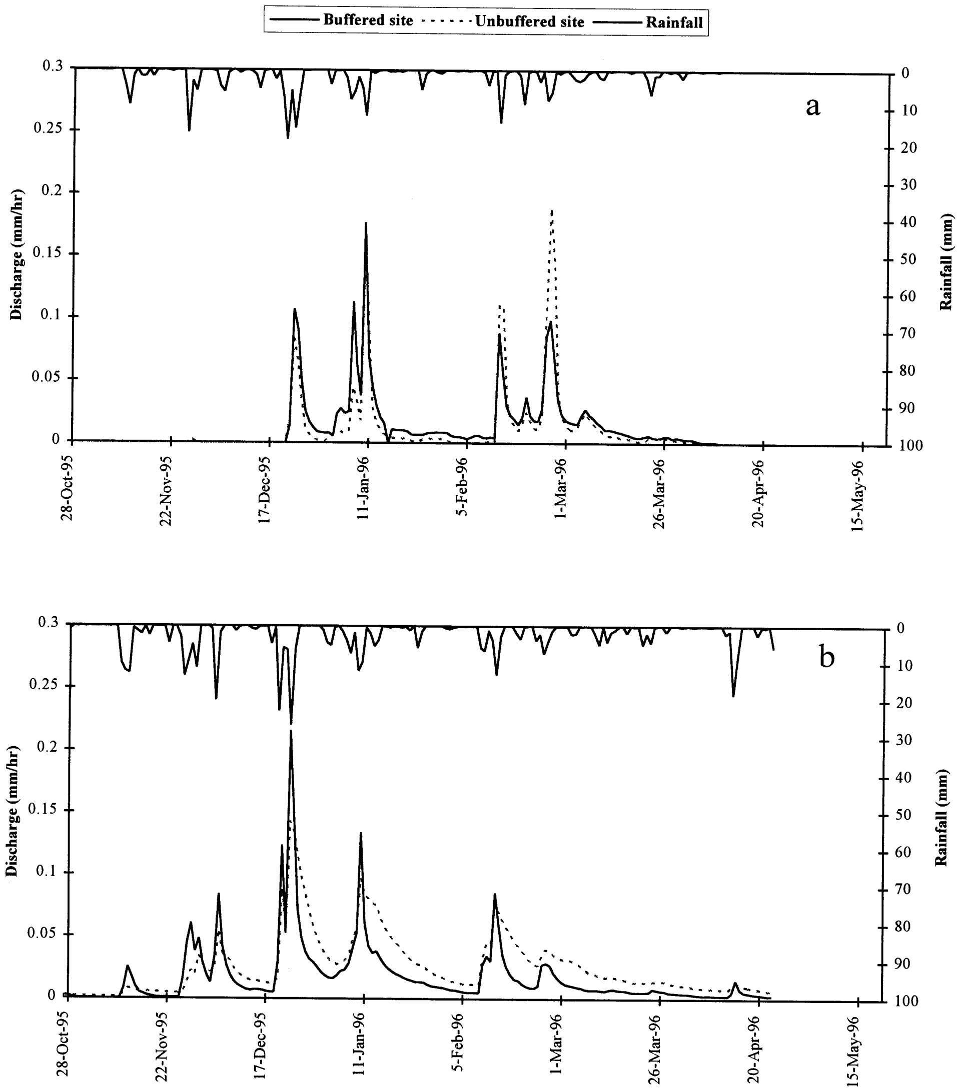

Fig. 3. Hydrographs for (a) Bedfordshire and (b) Oxfordshire buffered and unbuffered catchments (winter 1995–96).

Table 1
Maximum and mean recession time constants and times to peak flow in study catchments (winter 1995–1996 events)

Catchment	Recession time constant (h)		Time to peak flow (h)	
	Mean $\pm$ S.D.	Max	Mean $\pm$ S.D.	Max
Bedfordshire buffered	1.1 ± 0.4	1.6	6.7 ± 3.8	12.5
Bedfordshire unbuffered	0.7 ± 0.3	1.1	5.7 ± 3.3	10.5
Oxfordshire buffered	1.7 ± 1.4	4.6	3.1 ± 1.3	5.0
Oxfordshire unbuffered	4.1 ± 2.9	9.1	4.5 ± 1.6	6.5
Shropshire buffered	0.7 ± 0.3	1.3	3.3 ± 2.1	6.5
Shropshire unbuffered	0.7 ± 0.4	1.9	3.6 ± 2.8	9.0

S.D., standard deviation.

data implies that a fifth event probably occurred early in January 1996, when freezing conditions prevented a response by the floats in the stilling wells. The Bedfordshire catchments, like the Shropshire catchments had only negligible base flow during the excess winter rainfall period. Baseflow in the Oxfordshire catchments (Fig. 3b) was slightly higher, due to spring flow from the underlying limestone aquifer. However, baseflow was still small in comparison with the marked peak flow events observed in these streams. The Burlesdon and Stanway soils are characterised by shallow, saturated flow, and their impact in these catchments is significant, despite their limited extent.

In order to assess the speed of catchment response, hydrograph analysis was undertaken to determine the recession time constants and the time to peak flow (analogous to the time of concentration in these catchments). The recession time constant is taken as the inverse of the slope of the plot between the natural logarithm of the flow and time (Dougherty et al., 1995), assuming an exponential decay in the flow. Linear regression analysis confirmed that this assumption was valid in all of the study catchments. A smaller time constant indicates a faster system response. Time to peak flow is taken here as the time between peak rainfall and peak flow. Table 1 summarises the maximum and mean time constants and times to peak flow for the buffered and unbuffered study catchments.

The hydrographs of buffered and unbuffered catchments differ at all of the study sites. Most notably, the buffered catchment of the Shropshire site was often found to have considerably higher peak flows than the unbuffered catchment, particularly during very large storm events. This may be the result of more direct flow paths to the stream on this catchment when there is intensive rainfall. As cropping on both catchments was similar, this can be discounted as an explanation of this difference. The unbuffered catchment was 23% larger than the buffered catchment, which may imply a more rapid routing of water to the stream in the smaller catchment. The buffered catchment at the Oxfordshire site also had higher peak flows than the unbuffered catchment, although it also appeared to have somewhat faster recession times.

In the Bedfordshire catchments, recession time constants and times to peak flow tended to increase from the early part of the drainage period to the end. This may have been partly due to soil swelling, which can reduce the soil's bulk hydraulic conductivity. However, in such rapid response situations, the major influence on the shape of the hydrograph is likely to be differences in rainfall intensity. In the Shropshire catchments, the longest times to peak flow occurred in April 1996, when there appears to have been a small soil water deficit. If only the periods when the soil is at field capacity are considered, times to peak flow are typically only 2–3 h.

In all cases, the times to peak flow and recession time constants were less than 24 h, indicating short retention times for water in the catchments and little difference between the buffered and unbuffered catchments is noted. Such fast responses are not unusual for small headwater catchments with short streams (Kirpich, 1940; Bailey et al., 1980) and indicate short retention in the buffer strips. It is clear, and not surprising, that simply planting grasses on a buffer strip has little effect on the hydrology in these situations. The most likely explanation for the rapid response in these catchments is that preferential flow paths bypass the buffer strips.

3.3. Bypass flow mechanisms

The widespread installation of artificial field drainage, in the form of secondary mole drains connected with gravel backfill to underlying tile drains, has a strong influence on the hydrology of the Bedfordshire catchments. Such systems are known to give a rapid response (Dougherty et al., 1995). In the buffered catchment, 14 tile drain outfalls discharge directly into the measured stream. Studies during two of the drainage events showed that over 90% of the water flowing over the weir originated from the tile drains. Less than 10% of flow might thus be attributable to slow seepage. The recession time constants measured here are smaller than those reported on a similar mole drained clay soil in 1993–94 (Dougherty et al., 1995). The difference is explained by the very dry conditions of the summer of 1995, which left the soil in the study catchment in a highly fissured state, with macropores forming routing channels to the mole drains. Some pipe drains are also installed in the Shropshire catchments and in the Bursledon and Stanway soils of the Oxfordshire catchments, but mole drains are absent due to the structural instability of these silty soils. As a result, artificial drains are less important as a pathway for bypass flow on these sites compared with the Bedfordshire catchments. However, where subsoil waterlogging occurs on these sites, rapid preferential interflow takes place in the more hydraulically conductive topsoil, contributing to the flashy stream flow responses observed. The data clearly shows that the imposition of a grassed buffer strip alone cannot be expected to alter the hydraulic characteristics of the soil and so the subsurface hydrology, at least in the short term.

Measurements of nitrate–nitrogen concentrations in drain outfall discharge on the buffered Bedfordshire catchment confirmed that 90% of the nitrate–nitrogen load in the watercourse originated from the drains. Where mole drainage is present, solutes are quickly routed through the buffer strips from the topsoil to the subsoil where organic carbon levels are very low (approximately 0.1% in the case of clay subsoils in the Bedfordshire catchments) and denitrification is therefore unlikely. The widespread installation of artificial drainage on the less permeable soils also means that the anaerobic, waterlogged conditions suitable for denitrification seldom prevail in these headwater catchments. This situation is typical for catchments under arable agriculture as artificial drainage is necessary for timely mechanised cultivation where soils are prone to waterlogging. Although shallow lateral groundwater flow does occur in the Shropshire and Oxfordshire catchments, the opportunities for assimilation and denitrification are limited not only by the rapid rate of this subsurface flow, but also by the fact that this flow occurs only during the winter months. Soil temperatures are low at this time, and hence plant growth in the buffer strip is small. Although plant uptake of nitrate in the buffer strip is likely to be considerable during the summer months, there is little drainage from the catchment through the strip at this time, and it does not coincide with the peak period of nitrate loss from the land.

Concentrated overland flow is also known to act as a bypass mechanism that compromises the effectiveness of buffer strips in increasing retention times (Bailey et al., 1980). Surface run-off is uncommon in the Bedfordshire catchments because of the stable structure of the topsoil that results from the high clay content and abundance of free calcium carbonate. However, bypassing overland flow occurred in both the Shropshire and Oxfordshire catchments, where the silty and sandy loam topsoils are vulnerable to capping and sealing following rainfall impact, and flow is prone to become concentrated along compacted cultivation tramlines. In both cases, overland flow was observed and small rills were evident following major rainfall events.

Some slow seepage occurs on the Oxfordshire catchments, indicated by the measurable baseflow throughout the year. However, seepage takes the form of underdrainage, with direct infiltration of water from the arable land into the aquifer, from which it re-emerges as spring water at discrete points. Although the water in question has a long saturated flow path, it is through limestone rock, which has little denitrification potential. Although underdrainage is not a major contributor to peak flows on this site, this process also effectively bypasses the buffer strip.

It is clear that, for UK conditions, the buffer strip must provide a suitable environment for denitrification to take place, as the major nitrate fluxes occur during the winter months when grass growth is small and nitrogen uptake is also small. In these studies, the denitrification potential of the soils was considered to be adequate for treatment to take place. However, the rapid flows observed indicate that flow was shallow and bypassing, leaving little time for nitrification to take place. Generally, such rapid flow would indicate aerobic conditions predominate within the strip even in the winter.

Slowing down the rate of seepage through the strips would undoubtedly lead to wetter conditions and the opportunity for anaerobic conditions to develop in which denitrification can take place. If only 25% of the denitrification potential measured for the Oxfordshire buffered site were realised, then significant reductions in nitrate in the streams might be expected. It is clear, and perhaps unsurprising (see Hill, 1996), that simply sowing a grassed buffer strip adjacent to a stream in an arable headwater catchment has little effect on nitrate in the stream. What must be included is some engineered means of controlling the rate of seepage through the strip.

Stopping subsurface field drainage is one possibility. This might be achieved by restricting the flow from the drainage pipe but it is likely to be more satisfactory to install a weir in the stream, which backs up water through the drain and the strip. The reduction in hydraulic head across the strip that this would achieve would both create wetter conditions and increase retention times within it. Such engineering methods have been used successfully on lowland controlled drainage schemes (Evans et al., 1995). However, in headwater catchments such an engineering solution has considerable constraints on it due to the slopes of the stream and the surrounding land. Control structures would need to be placed at very frequent intervals in the stream in order to influence the whole length of the strip and costs of such a scheme may be prohibitive. An alternative may be to create open channels within the strip with a high water level into which drainage tiles discharge. In this way, water would be encouraged to wet the strip and to slow down that rate of flow through it.

4. Conclusions

(1) Previous research has demonstrated that riparian grass buffer strips have the potential to reduce levels of nitrate leaching from agricultural land into surface waters. However, as is apparent from this investigation, individual catchment hydrology is critical to the effectiveness of nitrate removing processes in the buffer strip.

(2) The need for diffuse pollution control is greatest in agricultural headwater catchments. These typically have very short water retention times, which in the study catchments were not significantly increased by the installation of buffer strips. This was thought to be due to the presence of preferential flow paths, which bypass the buffer strips. Depending on the site's soils and geology, these included channelling through artificial drainage systems, rapid overland flow and deep seepage. These processes limited the opportunities for vegetative assimilation of nitrogen and denitrification within the strips. The potential for nitrate uptake by the grasses was further reduced by low temperatures during the peak catchment drainage season. No substantial improvements in stream water quality were observed in the buffered catchments.

(3) The likely effects of soils and hydrology on buffer strip processes should be assessed on a case-by-case basis before recommending their installation. Modifications to buffer strip design may be possible to increase their effectiveness. Bypass flow could be minimised by careful engineering, and anaerobic conditions created by waterlogging at least part of the strip; although waterlogging may compromise the ability of the strip to trap sediment and have a detrimental effect on the arable land. The type of vegetation on the strip may also influence their effectiveness.

There is some evidence that forested strips may be more effective in removing nitrate during the winter months than grass strips (Haycock and Pinay, 1993) probably because such strips allow more organic carbon to accumulate, which is an energy source for anaerobic bacteria. The watercourse itself could be modified; for example, by installing retention structures to raise water levels, which would create lower hydraulic gradients across the strip, so slowing water flow and leading to wetter conditions within. Any engineering of riparian hydrology needs careful consideration as there may be adverse environmental impacts and, in any case, the effectiveness of such measures will depend on local soils and topography.

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