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Nitrate leaching from grazed grassland lysimeters: effects of fertilizer input, field drainage, age of sward and patterns of weather

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SUMMARY

Drained and undrained grassland lysimeter plots were established in 1982 on a clay loam of the Hallsworth series at a long-term experimental site in south-west England. The plots were continuously grazed by beef cattle, and received fertilizer at either 200 or 400 kg N ha^{-1} per annum to the existing permanent sward, or at 400 kg N ha^{-1} to a new sward, reseeded to perennial ryegrass following cultivation. Drainage water was monitored at V-notch weirs and sampled daily for the analysis of nitrate-N. Seven years of data are presented (five years for the reseeded swards). On the drained plots a large proportion of the rainfall was routed preferentially down large pores to the mole drains, whilst on the undrained plots, drainage was mainly by surface runoff. The average quantities of nitrate N leached per year were 38.5, 133.8 and 55.7 kg ha^{-1} from the old sward that received 200 and 400 kg N ha^{-1}, and from the reseed that received 400 kg N ha^{-1} fertilizer, respectively. Ploughing and reseeding resulted in a two-fold reduction in leaching, except during the first winter after ploughing, and twice as much leaching occurred after a hot, dry summer as after a cool, wet one. Nitrate concentrations in drainage from either drained or undrained plots were rather insensitive to rainfall intensity, such that concentration was a good predictor of nitrate load for a given drainage volume. The drainage volume determined the proportion of the leachable N that remained in the soil after the winter drainage period. Initial (peak) concentrations of nitrate N ranged, on average, from 55 mg dm^{-3} for the drained old sward that received 400 kg N ha^{-1} fertilizer, to 12 mg dm^{-3} for the undrained sward at 200 kg N ha^{-1} fertilizer input. Concentrations of nitrate N in drainage from similar, unfertilized plots rarely exceeded 1 mg dm^{-3}. The results suggest that manipulating the nitrate supply can lessen leaching and that the route of water through soil to the watercourse determines the maximum nitrate concentration for a given load.

INTRODUCTION

Concern about the leaching of nitrogen compounds from intensively-managed grasslands arose in the UK during the mid-1980s, when Ryden *et al.* (1984), Garwood & Ryden (1986) and Haigh & White (1986) presented clear evidence that nitrate concentrations in water draining from grazing land often exceeded the EC limit (50 mg dm^{-3}) for potable water, and that leaching could represent a serious loss of N. Hitherto, grassland agriculture had been considered environmentally benign compared with arable cropping (Lazenby, 1981; The Royal Society, 1983), mainly because the available evidence related to grassland that was cut, not grazed (Barraclough *et al.*, 1983), and despite indications that nitrate leaching could be substantial if fertilizer N was applied at rates

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exceeding 250 kg ha^{-1} to a free draining soil (Garwood & Tyson, 1973), and particularly after dry weather (Garwood & Tyson, 1977).

The main cause of the large losses of N under grazing is the large proportion of the N ingested by the animal that is rapidly recycled to the soil in excreta, mainly as urine (Whitehead, 1970). Moreover, these returns of N are highly concentrated and spatially variable (Ball & Ryden, 1984), thus giving rise to 'patchiness' in both herbage growth and potentially leachable inorganic N. Patchiness has important implications also for the choice of methodology to best estimate nitrate leaching: field-scale or catchment lysimetry (e.g. Haigh & White, 1986) is probably the most accurate, but is restricted to impermeable soils and cannot provide information on spatial variability. Point sampling of soil water by suction cups (Grossmann & Udluft, 1991) or by soil sampling (e.g. Wild, 1972) are the more widely-used alternatives but these methods must involve a degree of replication sufficient to accommodate the variability, and they can be criticized on the grounds that the N sampled may not be that which is subsequently leached.

An additional limitation to progress in this area is that large-scale field experimentation is necessary to investigate the interactive effects of the major edaphic, agronomic and climatic factors on N leaching from grazed grassland. Only a few experiments have been conducted therefore, over the last decade, from which to obtain data for the building and testing of predictive models. MacDuff *et al.* (1990) briefly described four such experiments and attempted to derive a generalized relationship between fertilizer N input and nitrate leaching. The relationship was rather poor, however, and only limited conclusions could be reached—that leaching was greater from sandy than from clay soils, and that grazing systems receiving less than 200 kg N ha^{-1} as fertilizer or through biological fixation are unlikely to produce mean annual concentrations of nitrate N in leachate that exceed the EC limit (11.3 mg dm^{-3}) for potable water. Of more importance in this respect however is a knowledge of how the actual concentration of nitrate varies with drainage volume. Few chemographs have been published of nitrate elution through structured grassland soils, and, because of the complication of 'preferential' water flow, attempts at simulation by models has met with only limited success (Barraclough, 1989*b*).

The Rowden Moor drainage experiment, sited in Devon, south-west England, has been in progress since 1982. Nitrate leaching from grazed grassland is monitored by catchment lysimetry, in relation to fertilizer input, sward type and the presence or absence of artificial field drainage. Other components of the N cycle under grazing are also measured. Preliminary data on N cycling from this experiment have been reported briefly (Garwood & Ryden, 1986; Scholefield *et al.*, 1988) and used as the basis of a predictive model (Scholefield *et al.*, 1991). We now report in detail some of the N-leaching data for the period 1983–1990.

EXPERIMENTAL

Site and soil

The experiment was set up on an old unimproved wet pasture, 7 km north of Dartmoor (SX 650995) sloping at between 5 and 10% from the 150 m contour to the east (Fig. 1). The existing sward, known to be at least 40 years old and with no history of arable cropping, has been used previously for extensive grazing with sheep and cattle. Thus the sward, dominated by *Agrostis* spp., *Holcus lanatus* and *Juncus* spp., and with less than 20% *Lolium perenne*, has received only small inputs of fertilizer prior to the experiment.

The soil is a clayey non-calcareous pelostagnogley of the Hallsworth series, overlying the clay shales of the Crackington Formation (Culm Measures), and is typical of much of the permanent grassland in the south-west of the UK. Details of the soil properties are given in Armstrong & Garwood (1991) and Scholefield *et al.* (1988).

Treatments

Twelve lysimeter plots, each 1 ha in area, were established in summer 1982, in two replicate blocks, and six treatments were applied. These comprised two levels of fertilizer N (200 and 400 kg ha^{-1}) on the existing sward, and one level (400 kg ha^{-1}) on a ploughed and reseeded sward (*Lolium perenne* cv. Melle), each with and without artificial field drainage. In 1987 a further two plots (each *c.* 0.6 ha)

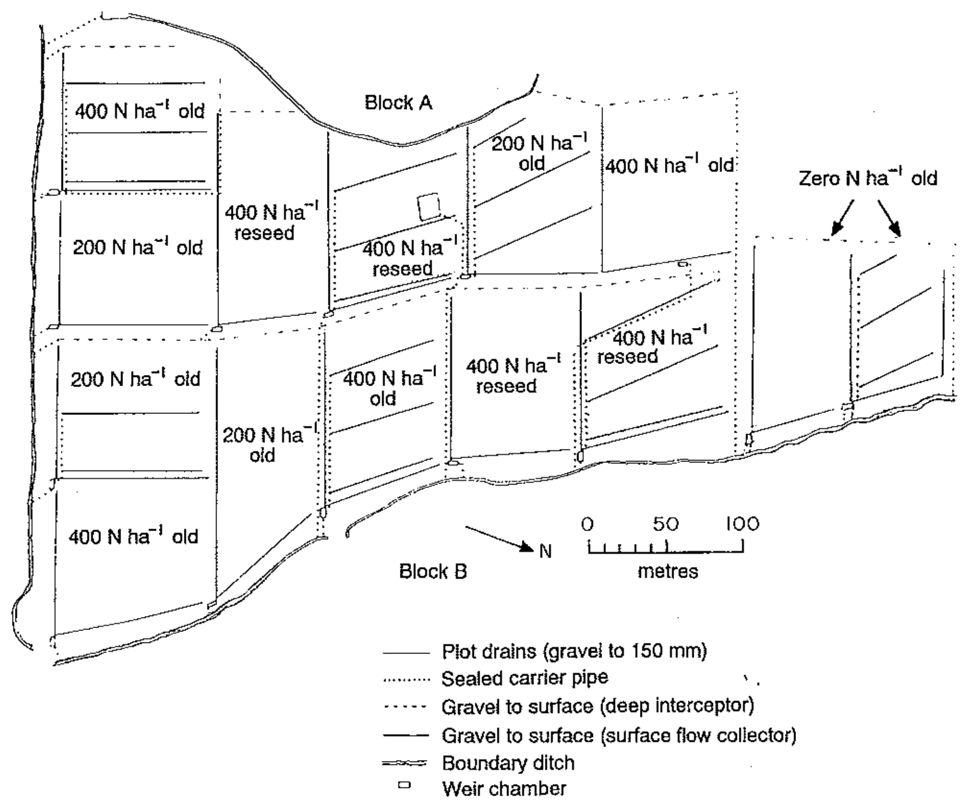

Fig. 1. Layout of North Wyke drainage experiment.

were added to the experiment, one drained and the other left undrained, on the existing sward, to be managed with no input of fertilizer N. Three years of leaching data from the latter are included in the present paper.

Drainage was by mole drains, drawn up and down the slope at 2 m spacing and 55 cm deep, crossing permanent pipe drains (> 100 mm diameter) at 40 m spacing and 85 cm deep, with permanent backfill to within 30 cm of the surface. At the downslope boundaries of each plot, surface water interceptor drains were installed at 30 cm depth with gravel backfill to the surface, and at the upslope boundaries, deep interceptor drains with backfill were installed to divert extraneous water. Thus, each plot was hydrologically isolated, except for deep seepage which was thought to be negligible for a subsoil with such a low hydraulic conductivity (Armstrong & Garwood, 1991), and the likelihood of deep seepage into the plots by water moving downslope under pressure was reduced by the placement of some extra intercepter drains. This drainage design was adopted to accommodate estimated peak discharges up to 28 dm^3 ha^{-1} s^{-1}, reflecting the hydrology of the site, rather than the transmissive properties of the pipe-work.

Both drainflow and surface water flow from each plot were measured using the head recording system described by Talman (1983) and a weir plate for a 1/2 90° weir (British Standards Institution, 1965). Further details of the hydrological monitoring procedures and soil conditions are given in Armstrong & Garwood (1991).

Nitrogen fertilizer was applied to the plots as ammonium nitrate in nine equal amounts of 22.2 or 44.4 kg ha^{-1}, starting in March of each year as soon as ground conditions permitted, and subsequently at intervals of 3 weeks from mid-April. Lime (5 t ha^{-1}), P (32 kg ha^{-1}) and K (60 kg ha^{-1}) were applied to the reseeded swards at their establishment, but no P or K was applied to the existing swards until the following spring (1983) when 5 t ha^{-1} of lime was also applied to these swards. Thereafter, each spring, 25 kg ha^{-1} of P and 50 kg ha^{-1} of K were applied to all plots. The pH of the 0–10 cm soil layer was maintained above 6.0 by addition of lime to all plots in spring 1986.

The swards were continuously grazed each year by beef steers averaging 295 kg liveweight at turnout. Steer numbers were adjusted on a day-to-day basis to maintain a sward height between 50 and 60 mm (rising-plate sward stick; Holmes, 1974). The start and the end of each grazing season was determined by the bearing strength of the ground measured with a hand-held shear vane (Pilcon Engineering, Hemel Hempstead) and by the amount of herbage on offer. The effects of the treatments on herbage and animal production were described by Tyson *et al.* (1992).

Assessment of nitrate leaching

During the winter drainage period, water samples were collected once per day (more frequently during intense storm events) from the outflow of each weir. The samples were stored frozen to be analysed for ammonium and nitrate by standard automated colorimetry (Hendriksen & Selmer Olsen, 1970). Concentrations of ammonium were less than 0.6 mg dm^{-3}, so only nitrate concentrations are reported.

Automated flow-proportional samplers (Electronic and Technical Services Ltd, New Brighton, UK) were installed during the winter of 1984, but these proved unreliable for long-term continuous operation and their use was discontinued. While they were operational (18 October to 31 December 1984), however, a comparison was made between both the daily and longer-term mean nitrate concentrations obtained by automatic composite sampling and manual spot sampling. Daily values were very closely matched, with no systematic differences, such that for means of up to 38 days the two procedures gave values differing typically by less than 2%. The reason for good correspondence was that the concentration of nitrate in water draining a given plot varied only slightly from day to day, such that 20 mm rainfall in 24 h resulted typically in 10% variation during the day. It was concluded from these comparisons that only for storm events of intensity greater than 35 mm d^{-1} would manual sampling be unacceptably inaccurate (seven such events only were recorded from 1983–1987).

The quantity of nitrate N leached was calculated weekly by multiplying the average concentration by the drainflow. Values for drainflow were obtained either from the hydrographs, or assumed to be equal to rainfall minus evapotranspiration. Both values are subject to error in that any nitrate leaking through the plot boundaries may not be at the same concentration as that monitored at the weir. Using hydrographs will tend to underestimate leaching because water collection will not be completely efficient, but assuming that drainflow equals rainfall minus evapotranspiration means that the values for drainflow and nitrate concentration are displaced in time by the residence time of the rainwater. During the weekly assessment period this would have been unlikely to cause a large error because of the slow rate of change of nitrate concentration.

Meteorological recording

A Meteorological Office station (North Wyke, Station 8836) is situated within 2 km of the experimental site and data obtained for rainfall and evapotranspiration measured there were considered representative of the experimental site. Potential transpiration was estimated daily for evaporation of water from a tank using seasonal correction factors (Penman, 1952) and the potential soil water deficit was assumed equal to potential transpiration minus rainfall since field capacity.

Statistical treatment of results

As with all large-scale field experiments carried out with finite resources, a compromise had to be reached between the size of each plot (the average number of grazing animals per plot), the number of treatments and the degree of replication. The small degree of replication made it desirable to bulk 'sward type' and 'fertilizer-N rate' within 'drained' and 'undrained' treatments for each year, to maximize the number of degrees of freedom within 'drainage'. Thus an analysis of variance using Genstat (Genstat 5 Committee, 1987) could be performed on the quantity of nitrate leached annually with 11 degrees of freedom, of which 1, 2 and 8 were allocated to drainage, agronomic treatment (sward, fertilizer) and residual respectively.

RESULTS

Patterns of weather and water balances

Tables 1 and 2 show that there were large contrasts in patterns of weather over the experimental period: the summers of 1983, 1984 and 1989 were warmer and drier than average, resulting in large soil water deficits, whereas the 1985, 1986 and 1988 summers were cooler and wetter, resulting in smaller than average soil water deficits. The two driest summers (1984 and 1989) were followed by the two wettest winters, when the largest volumes of drainage were recorded.

Once drainage had stopped in spring it did not normally restart until late autumn, and it was only in August 1985 and September 1986 that the soil returned temporarily to field capacity during the grazing season. During the 1988–89 winter the drainage volume was the smallest but the drainage duration was the longest, whereas during the following winter the drainage volume was the greatest but the duration shortest.

Table 1. Monthly mean soil temperature (°C) at 10 cm. Measurements taken daily at 09.00 h

	1983	1984	1985	1986	1987	1988	1989
Jan	5.9	4.2	1.5	4.2	1.8	5.5	5.8
Feb	2.1	3.6	3.0	0.6	3.6	3.9	5.3
Mar	5.8	3.9	4.0	4.1	4.5	5.8	6.6
Apr	6.1	6.9	7.7	5.1	8.6	7.5	6.6
May	9.6	9.8	10.6	9.7	10.2	11.1	13.1
June	14.1	14.9	13.2	14.0	13.0	14.7	14.4
July	18.7	16.0	15.8	15.8	16.0	14.3	17.5
Aug	16.6	16.0	13.8	13.1	15.5	14.3	15.5
Sept	13.2	12.9	13.4	10.8	13.3	12.9	13.3
Oct	10.4	10.4	10.6	10.4	9.2	9.9	11.4
Nov	8.2	7.2	4.9	7.3	6.9	6.0	7.6
Dec	5.7	5.4	6.6	5.9	5.7	7.2	5.4
Mean	9.7	9.3	8.8	8.4	9.0	9.4	10.2

Table 2. Annual rainfall, maximum potential soil water deficit (SWD), drainage volume and duration

Year (1 April–31 March)	Rainfall (mm)	Maximum potential SWD (mm)	Maximum potential SWD Date reached	Drainage volume (mm)[a] 1	Drainage volume (mm)[a] 2	Drainage duration First autumn	Drainage duration Last spring
1983–84	1144	268	1 Aug	555	420	28 Nov	8 Apr
1984–85	970	308	21 Aug	655	441	18 Oct	16 Apr
1985–86	851	140	26 July	410	270	8 Nov	26 Apr
1986–87	1091	85	30 July	630	502	20 Oct	9 Apr
1987–88	1103	198	2 Oct	620	375	10 Oct	4 Apr
1988–89	930	104	26 June	510	379	27 Sept	1 May
1989–90	1199	347	8 Sept	730	—	28 Oct	2 Mar
Mean	1041	207	17 Aug	587	—	24 Oct	9 Apr

[a]1. Calculated from rainfall minus evapotranspiration once field capacity was reached.
2. Mean of volumes measured through V-notch weirs.

The proportion of the drainage volume, calculated as rainfall minus evapotranspiration, that was monitored on average through the weirs ranged from 61% in 1987–88 to 80% in 1986–87 (Table 2). The discrepancy in the water balance and its variation reflects the integrity of the plots as water catchments, the efficiency of the systems for monitoring drainflow and the accuracy in estimating evapotranspiration. It appears that these factors were not related to drainage volume or to the age of the mole drains, which were re-drawn in 1987 without improving the average water balance.

Nitrate leaching

The effects of fertilizer, drainage and annual patterns of weather on nitrate leaching are given in Table 3 and Figs 2 and 3. The main body of results and the relationships derived from them have been calculated assuming drainflow equals rainfall minus evapotranspiration. Continuous hydrographs were obtained for all weirs only during the winters of 1983–84 and 1984–85; the nitrate leaching data calculated on this basis are included in Table 3 for comparison. Over these two winters, nitrate leaching calculated from hydrograph data was, on average, 76% and 67% respectively of that calculated from meteorological data. The variability between replicate plots was apparently smaller with hydrograph-based calculations however, which led to smaller standard errors.

Table 3. Losses of nitrate-N by leaching (kg ha^{-1} a^{-1}), (a) calculated assuming drainflow = rainfall less evapotranspiration, (b) calculated assuming drainflow = water monitored through weirs (2 years only)

Drainage year	1983–84		1984–85		1985–86	1986–87	1987–88	1988–89	1989–90
Treatment	a	b	a	b	a	a	a	a	a
200 N old sward	39.8	28.4	51.9	44.0	41.0	21.4	39.5	20.6	54.4
400 N old sward	135.7	92.5	186.0	129.4	144.0	90.4	141.7	66.8	172.1
400 N reseed	66.6	41.6	59.7	43.5	48.0	35.4	68.6	—	—
SED	16.7	11.2	17.9	8.4	25.3	15.3	11.9	15.1	21.4
Level of significance[a]									
fertilizer	*	**	***	***	*	*	***	NS	*
swards within 400 N	*	**	**	***	**	*	***	*	**
Undrained	32.6	20.5	61.9	41.6	42.0	19.2	39.5	5.6	57.7
Drained	113.6	81.4	136.6	115.0	113.0	79.0	127.0	81.8	168.8
SED	13.6	8.4	14.6	6.2	20.7	12.5	9.7	12.4	17.5
Level of significance[a]	***	***	**	***	*	**	***	**	**
Zero N undrained (unreplicated)							2.6	1.2	3.4
Zero N drained							3.0	4.0	8.4

[a]*, $P<0.05$; **, $P<0.01$; ***, $P<0.001$; NS, not significant ($P>0.05$).

During intense storms, flows were recorded at each weir receiving surface water from the drained plots. The nitrate concentration in this water was typically less than 10% of that measured simultaneously at each companion weir receiving water from the drainage system, such that nitrate leaching by surface run-off from each drained plot was less than 3% of the annual total. Consequently, these small contributions have been incorporated with the annual totals in Table 3.

Increasing the input of fertilizer N from 200 to 400 kg ha^{-1} on the old swards increased the N lost by leaching 3-fold on average, with the equivalent of 19% and 33% of the fertilizer input lost at 200 and 400 kg ha^{-1} respectively. The leaching losses from the reseeded swards were less than half those from the old swards, with the equivalent of only 14% of the fertilizer lost during 1983–87. There would have been a large quantity of nitrate leached from the ploughed swards during the 1982–83

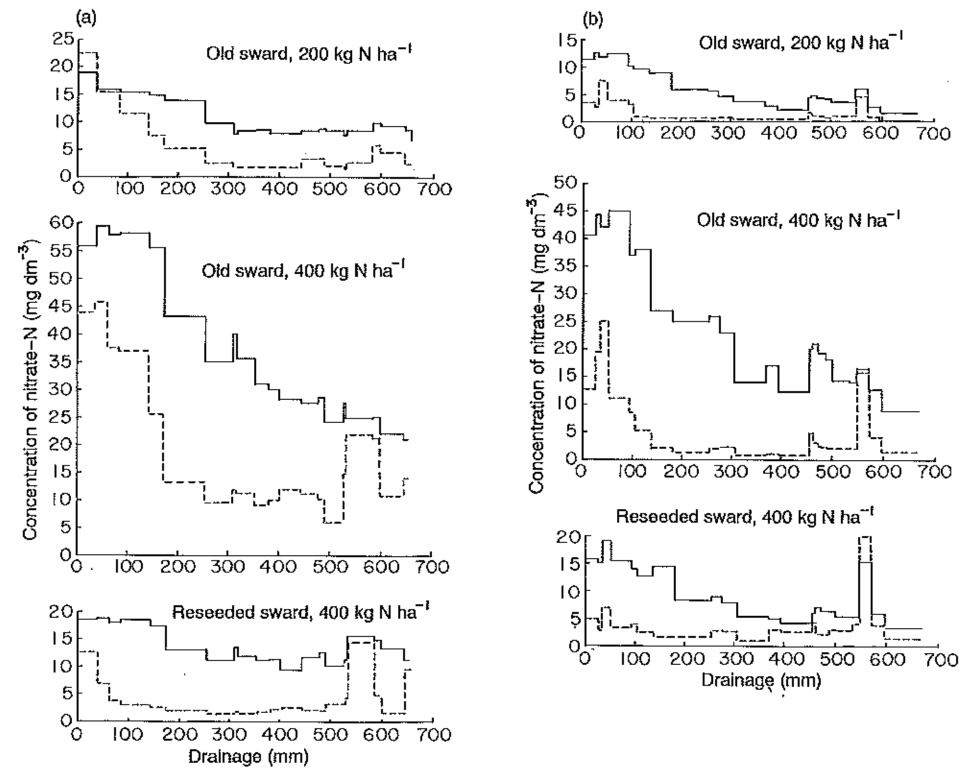

Fig. 2. Patterns of nitrate leaching from drained (thick line) and undrained (dashed line) plots in (a) 1984: a high leaching year, and (b) 1986: a low leaching year.

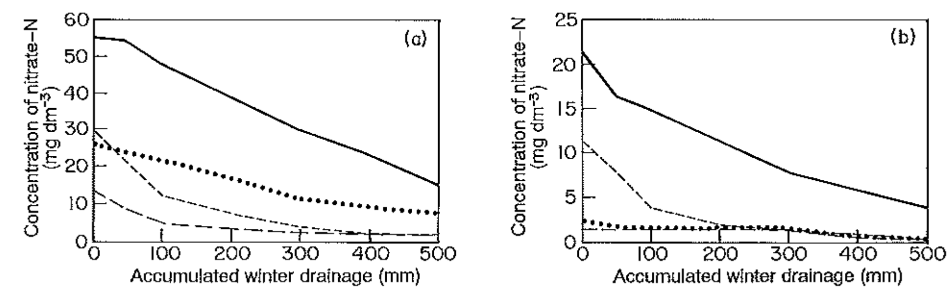

Fig. 3. Patterns of nitrate leaching averaged over the number of years for each treatment: (a) all treatments at 400 kg N ha^{-1} fertilizer input, —— old sward drained, – – – – old sward undrained, · · · · · · reseed drained, — — — reseed undrained, and (b) treatments on the old sward at zero and 200 kg N ha^{-1} fertilizer input, —— 200 kg N ha^{-1} drained, – – – – 200 kg N ha^{-1} undrained, · · · · · · zero N drained, — — — zero N undrained.

winter but this was not assessed. Land drainage, in contrast to reseeding resulted in a 3-fold increase in nitrate leaching, averaged over sward type and over the experimental period. The size of the effect of drainage was consistent over sward type and quantity of fertilizer, although such information is obscured by the way the data were analysed and presented in Table 3.

The effect of weather on leaching was of the same order as that of fertilizer, sward age or drainage status with, more than a 2-fold range between data obtained in 1984 and 1986. Detailed chemographs of nitrate leaching during these contrasting years (Fig. 2a, b) show broadly similar features however, with nitrate concentrations in 1984 starting and remaining at least 50% higher than in 1986. An important feature of the chemographs is that they show direct loss of fertilizer N applied for early spring growth. This loss, which was mainly due to 'surface runoff' from the undrained plots, represented less than 30% of the N applied, but for the undrained reseeded plots, this loss was a substantial proportion of the annual loss through leaching.

One difference between the chemographs for the two years is that in 1984, a year with large nitrate leaching losses, the initial nitrate concentrations in leachate from drained and undrained plots were much closer than they were in 1986, a year with much less nitrate leaching. In both years and over all treatments however, the attenuation in concentration with drainage volume was greater for the undrained than for drained plots, an effect best demonstrated in Fig. 3. This figure also shows that, at 400 kg N ha^{-1}, the undrained reseeded treatment was the only one that gave nitrate concentrations in the drainage water that were below the EC limit for most of the year. The nitrate concentrations in drainage from the old sward given 400 kg ha^{-1} of N remained above this limit. Concentrations of nitrate N in drainage from plots that received no fertilizer N rarely rose above 1 mg dm^{-3}, with total losses of N of 2.4 and 5.1 kg ha^{-1} from the undrained and drained treatments respectively averaged over the three winters of 1987–90.

DISCUSSION

Factors affecting nitrate leaching: nitrate supply

Comparison of the leaching data (Table 3) with those from other experiments, summarized by Vinten *et al.* (1991), substantiates the earlier conclusions of Ryden *et al.* (1984) and Steele *et al.* (1984) that nitrate leaching losses from grazed grassland are greater than those from grassland that is cut, and are commensurate with amounts from arable systems, given similar amounts of fertilizer. The results show also that the quantity of fertilizer supplied, the age of the sward, drainage status and the weather can each exert a large influence on the amount of N leached, so that attempts to establish precise and generally applicable relationships between any one of these and nitrate leaching are unlikely to be successful (e.g. fertilizer input; Macduff *et al.*, 1990). In a country like The Netherlands, with a small range of soil types, it may be possible to hope to derive such a relationship for average weather conditions (e.g. Van der Meer & Meeuwissen, 1989), but a similar relationship derived from the data of the present experiment (Fig. 4) should be expected to be site-specific and accurate only for 'average' weather conditions.

The general form of this relationship was presented by Steenvoorden *et al.* (1986) as a curve with a gradient that increases continuously as a greater proportion of each increment of fertilizer is leached. Barraclough *et al.* (1992), however, obtained data on nitrate leaching for a number of grazed grassland sites in the UK that showed what they describe as a 'breakpoint' in the relationship, indicating the possibility of identifying a threshold fertilizer input above which much greater leaching losses would occur. Unfortunately, as these authors suggest, the breakpoint, if it exists, is also likely to be site-specific and year-dependent.

Preliminary reports of the N-cycling data from the present experiment (e.g. Garwood & Ryden, 1986; Jarvis *et al.*, 1989) suggested an 'interdependence' between leaching and denitrification that might help to explain site specificity, according to differences in soil wetness. A quantitative interdependence is not however supported by the longer-term data on annual nitrate leaching (presented here) and denitrification measured over 4 years (Scholefield *et al.*, 1988) from drained and undrained plots. Indeed, the modest 30% reduction in annual N loss through denitrification due to drainage did not balance the approximately 300% increase in nitrate leaching, and could, moreover, be accounted for by the enhanced plant uptake of N in the drained plots (Scholefield *et al.*, 1988). Clearly, other explanations for the effect of field drainage must be sought.

In a mass-balance approach to modelling the annual N cycle under grazing, Scholefield *et al.* (1991) showed that the larger the total input of mineral N to the soil (from fertilizer, mineralization, urinary returns and atmospheric deposition) the larger the proportion that is lost to the environment through leaching, denitrification and the volatilization of ammonia. Simulations of the relationship between the input of fertilizer N and nitrate leaching, for different combinations of site

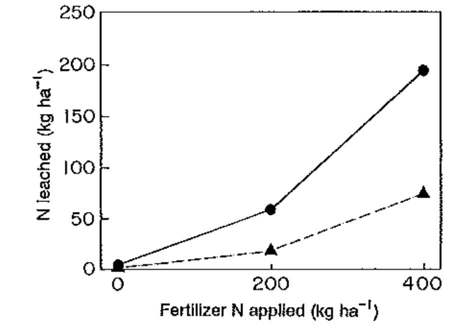

Fig. 4. Relationships for drained (●) and undrained plots (▲) between fertilizer N applied and the average amounts of nitrate N leached.

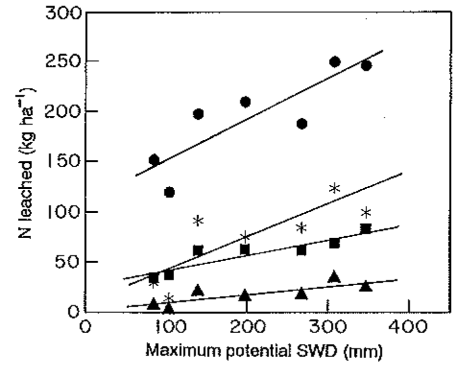

Fig. 5. Linear regressions of nitrate leaching from the old swards on the maximum potential soil water deficit.

●, 400 N drained, $y = 112.9 + 0.3\,x$, $r^2 = 0.73$;
*, 400 N undrained, $y = 10.3 + 0.31\,x$, $r^2 = 0.67$;
■, 200 N drained, $y = 30.6 + 0.13\,x$, $r^2 = 0.64$;
▲, 200 N undrained, $y = 2.2 + 0.07\,x$, $r^2 = 0.50$.

characteristics, did not reveal the existence of a 'breakpoint' on which to base 'safe' fertilizer recommendations. In this approach, mineralization, as determined by soil aerobicity, water content, temperature, organic matter and the C:N ratio of plant residues, was identified as the main factor causing site specificity, and the one most influential on nitrate leaching, after fertilizer input.

In another investigation, annual net mineralization was measured continuously from June 1988 to June 1989 in the 0–20 cm soil layer of four of the field plots used in the present study. Values for this period were 63, 146, 141 and 279 kg N ha^{-1} for the undrained and drained treatments at zero fertilizer and the undrained and drained treatments at 400 kg N ha^{-1} fertilizer input, respectively. These data represent an independent assessment of the effects that soil physical conditions (as mediated through drainage) and the N content of plant residues may have on the supply of soil mineral N, and on N loss through leaching. However, neither the mass-balance calculations, nor the concordant results of the direct measurements of mineralization rule out the possibility that substantial amounts of either water-soluble or colloidal organic N may have been leached, particularly from undrained plots.

Similarly large differences in annual net mineralization of N might be expected in soil from old and reseeded swards (Tyson *et al.*, 1991). Measurements of soil properties on the different plots of the site reported in 1988 (Scholefield *et al.*) reveal that ploughing and reseeding of this grazed pasture eventually resulted in reduced aerobicity and organic matter content of the 0–10 cm soil layer, both factors being conducive to lower rates of mineralization.

On commercial farms fertilizer N would probably be applied less frequently than in this experiment, and with a greater proportion of the total applied earlier in the season. In order to assess the applicability of the results therefore, it is important to consider whether the pattern of fertilizer application is likely to influence the amount of nitrate leached. Titchen & Scholefield (1992) showed that by applying fertilizer strategically when there was greatest demand by the herbage it was possible to partition more of the applied N to the product (beef), and so reduce the residue of potentially-leachable soil N in the autumn. Model calculations indicate however (Scholefield *et al.*, 1991) that, because of the poor efficiency of N utilization by the animal, a 25% increase in the N partitioned to product would result in a less than 2% reduction in the N that remained cycling in the system. Although a greater proportion of this N would be in the organic form, it could be re-mineralized during autumn and winter to negate any benefits in terms of reduced leaching (Titchen & Scholefield, 1992). It may be assumed therefore, that the pattern of fertilizer application is likely to have little impact on the amount of N leached, except if conditions in autumn and winter are unfavourable for mineralization, the rate of denitrification is influenced, or if fertilizer is applied immediately before the onset of drainage.

The effects of weather on nitrate leaching are very complex and, as with the effects of soil physical conditions, the complexity can be reduced conceptually by considering the effects on nitrate supply separately from those on nitrate transport through the soil. Weather influences nitrate supply by affecting the interactions between plant uptake, mineralization, nitrification and denitrification. Whilst detailed mechanistic models are necessary to understand fully and predict the outcome of these interactions, empirical approaches may be adopted to estimate broad trends and also to evaluate the relative importance of the weather's influence on nitrate supply and transport. For example, the influence of weather over the growing season might be evaluated by relating leaching in any year to the maximum potential soil-water deficit attained over the previous summer. Hot dry years should give rise to more nitrate leaching since both denitrification and plant uptake of N would be lower than during cool wet years. Moreover, the longer the period of hot, dry weather, the greater would be the rate of mineralization of N on re-wetting. This effect is thought due to the rejuvenation of microbes after a partial sterilization of the soil (Birch, 1959).

Figure 5 shows linear regressions of nitrate leached on maximum soil-water deficit (Table 2) for the four 'old sward' treatments, from which it may be inferred that summer weather had a major effect on the supply of leachable nitrate. Figure 5 indicates 'weather dependent' ranges of leaching of about 30 and about 100 kg N ha^{-1}, at 200 and 400 kg N ha^{-1} fertilizer input respectively, which is in line with effects of weather observed by Garwood & Tyson (1977) and Jordan & Smith (1985). These relationships demonstrate the large scope for improving the efficiency of N use in dry years by withholding fertilizer input, an approach adopted by Titchen & Scholefield (1992), or through irrigation to improve plant uptake, as advocated by Steenvoorden (1987).

Factors affecting nitrate leaching: nitrate transport

The relative importance of the mechanisms of water flow and nitrate transport as opposed to nitrate supply in determining the quantity of nitrate leached is difficult to ascertain but has become a much debated issue. The argument that the mechanism of transport is unimportant because with sufficient drainflow all nitrate would eventually leach is not supported by recent measurements of Trudgill *et al.* (1991), who concluded that nitrate losses from a small drainage basin in Devon were transport-limited and largely independent of variations in supply.

The transport of solutes like nitrate through structured soils is complicated by the phenomena of 'preferential' and 'by-pass' flow (Barraclough, 1989*a*). A proportion of soil water is held in relatively immobile zones and interaction of this water with water draining more rapidly through cracks and macropores is limited to an extent dependant upon numerous factors, including pore geometry and the intensity and duration of the rainfall. Preferential flow has important consequences for nitrate leaching in that under wet soil conditions nitrate is held back in relatively immobile water and leaches at a smaller concentration than it otherwise would by convective–dispersive flow. Under drier conditions however, by-pass flow of intensive rainfall can leach large proportions of nitrate fertilizer applied to the soil surface (Barraclough *et al.*, 1983; Smaling & Bouma, 1992). Such diverse effects prove difficult to simulate by highly mechanistic models partly because of the large temporal and spatial variability of the key parameters (Barraclough, 1989*b*).

A high degree of preferential flow would be expected in the mole-drained clay loam of the present experiment, an expectation which has been verified in tracer experiments (Hallard & Armstrong, 1992), and is confirmed by comparison of Fig. 2 with chemographs of nitrate leaching from a Wickham series clay loam (Barraclough, 1989*b*) and of chloride leaching through the Rothamsted drain gauges (Addiscott *et al.*, 1978).

The mechanism by which solutes are transported downslope through surface layers in 'run-off' from the undrained plots can only be speculated upon. Thus it may be appropriate to consider run-off as an example of by-pass flow (the ground surface being the macropore), which becomes increasingly extreme with increasing slope. The highly skewed distribution of nitrate concentration over the drainage volume (Figs 2 & 3) would support the validity of this model. Whilst the loss of spring-applied fertilizer was greater from the undrained plots of the present experiment, less than

10% of the fertilizer N applied to grassland run-off lysimeters on an adjacent site in February was leachable (Scholefield & Stone, 1992). This indicates that a high degree of protection from by-pass flow was afforded by rapid diffusion into the micropores of the wet soil. An important aspect of leaching by run-off however, is that even a small nitrate load would give rise to a larger peak nitrate concentration relative to that produced by preferential flow to the mole drains.

The difference between the average elution profiles of the drained and undrained plots (e.g. Fig. 3a) do not necessarily reflect differences in the mechanisms of nitrate transport, since differential rates of mineralization, denitrification and plant uptake may also have been responsible. Nitrate concentration in leachate from both drained and undrained plots was rather insensitive to drainage (rainfall) intensity, giving rise to the possibility, in each case, of deriving empirical relationships between nitrate concentration, nitrate leached (load) and drainage volume. Thus, it is possible to predict nitrate leached from the drained plots, for a given drainage volume, from its initial (peak) concentration (see Figs 3a & 6). A similar possibility is indicated for the undrained plots but with the slope of the regression line ($y = 0.71 + 0.58x$, $r^2 = 0.96$) of initial nitrate concentration on the total amount leached being markedly steeper. Alternatively, such relationships could be used in the development of mass-balance models (e.g. Scholefield *et al.*, 1991) for the estimation of maximum nitrate concentration from annual load.

Figure 7 shows also that, with winter drainage of 400 mm, over 80% of the potentially-leachable nitrate might be expected to be leached from the drained plots, and Fig. 4b indicates that an even greater proportion would be leached from the undrained plots. Only after exceptionally dry winters therefore will there be substantial residues of nitrate in well-structured grassland soils in the west of the UK. In the east of the UK, and in many regions of continental Europe with clay soils, where there is typically less than 200 mm of drain flow during winter, less than 50% of the leachable N will

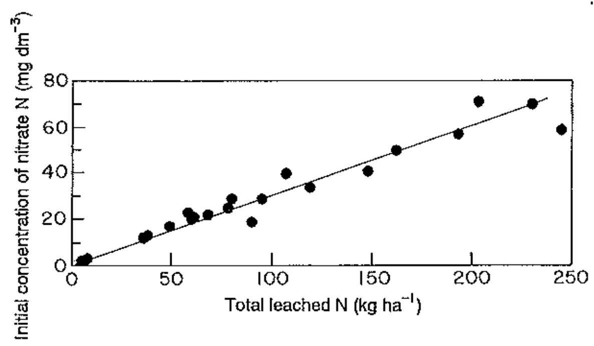

Fig. 6. Linear regression of the initial (maximum) concentration of nitrate N in drainage on the total amount of N leached from the drained plots ($y = 2.74 + 0.28\,x$, $r^2 = 0.97$).

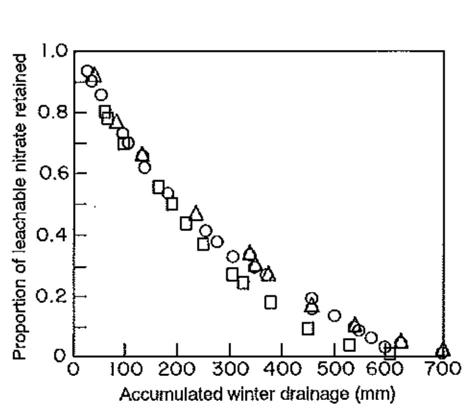

Fig. 7. Relationship between the proportion of the annual total of leached nitrate retained and the accumulated drainage volume for the drained plots: ○, 1986–87; □, 1987–88; △, 1989–90.

be leached. In these regions substantial accumulation of nitrate might be expected where intensive grazing is practised on 'permanent' swards.

CONCLUSIONS

1. Large amounts of nitrate were leached from grazed grassland receiving fertilizer N at 200 or 400 kg ha^{-1}, with nitrate concentrations frequently in excess of the EC limit for drinking water. Very small amounts were recorded at zero fertilizer input.
2. Nitrate leaching was greatly influenced by the supply of fertilizer N, drainage status, sward age and patterns of weather, such that attempts at prediction of leaching from fertilizer alone are likely to be fruitless.
3. Installation of an efficient field-drainage system resulted in a three-fold increase in nitrate leaching, which was probably due mainly to enhanced mineralization during the grazing season and possibly over autumn and winter.
4. Ploughing and reseeding at 400 kg N ha^{-1} fertilizer input greatly decreased nitrate leaching relative to the old sward, but there would have been a large loss of mineralized N over the first winter following reseeding, that was not monitored in the present experiment.
5. Patterns of summer weather greatly influenced nitrate supply such that there was twice as much leached after a hot dry summer as after a cool wet one. The amount of winter drainage had a rather small effect on leaching, despite a range of 400–700 mm.
6. The intensity of rainfall during the winter drainage period had only a small influence on the concentration of nitrate in leachate from either drained or undrained plots. This demonstrates the possibility of deriving robust empirical relationships between concentration, nitrate load and drainage volume, in order to predict the outcome of preferential flow in structured soils.
7. Surface run-off from the undrained plots may be an example of by-pass flow, since for a given load of soil nitrate, the initial concentration was greater than in leachate from the mole-drained plots. Whilst this result has important consequences in relation to the environmental impact of artificial field drainage, the study as a whole also points to the need for further investigation into the mechanism of the transport of pollutants in run-off.

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