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Seasonal fluctuations in the mineral nitrogen content of an undrained wetland peat soil following differing rates of fertiliser nitrogen application

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ABSTRACT

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Fluctuations in soil mineral nitrogen (N) caused by repeated applications of fertiliser N were investigated over 3 years in wetland hay meadows in Somerset. Swards were cut for hay after 1 July each year and the aftermath grazed continuously with beef cattle until October. Several N treatments between 0 (N-0) and 200 kg N ha^{-1} $year^{-1}$ (N-200) were applied in two equal dressings each year. Denitrification rates were estimated over two consecutive autumn–winter periods, using a soil core incubation method.

A high proportion of soil N was in nitrate form for most of the year and therefore at risk of loss from both leaching and denitrification. Estimated losses of total soil mineral N (ammonium plus nitrate) between October and March averaged 17.8 kg N ha^{-1} at N-0, 30.0 kg N ha^{-1} at N-100 and 88.2 kg N ha^{-1} at N-200 for the 3 years 1987–1988, 1988–1989 and 1989–1990. Denitrification losses were significantly related to the amount of applied N in both 1988–1989 and 1989–1990, but only accounted for a small proportion of the total N loss from N-200 soils. Mean leaching losses in autumn–winter were estimated at about 5 kg N ha^{-1} for N-0, 17 kg N ha^{-1} at N-100 and 67 kg N ha^{-1} at N-200. Further losses were suspected during the 1988 summer, when prolonged rain following N application resulted in flooding in early October.

These estimates are based solely upon indirect measurements of N loss, but they do give a good indication of the scale of losses incurred. It is concluded that these soils are particularly susceptible to N leaching, as a result of a combination of high water-tables and apparently rapid conversion of ammonium N to nitrate (nitrification). Annual fertiliser rates should be kept below 100 kg N ha^{-1} to avoid leaching risk. Taking an earlier cut for hay or silage could increase N efficiency and reduce losses of mineral N into the environment, although early cutting might be unacceptable because of its adverse effect on the breeding success of wading birds.

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INTRODUCTION

The low-lying peat moors of the Somerset Levels are one of the few remaining extensive wetland habitats in Britain. They consist mainly of meadows with high water-tables, lying within a close network of ditches and watercourses. These meadows are of great ecological interest, not only for the diversity of flowering plants they support, but also because they provide valuable breeding and overwintering sites for wading birds. Some 6500 ha, in total, of these moors have been designated as Sites of Special Scientific Interest, with their management strictly controlled by English Nature (formerly the Nature Conservancy Council) in return for negotiated financial compensation. A much wider area (30 000 ha) has been declared an Environmentally Sensitive Area (ESA) in which a voluntary scheme operates (Her Majesty's Stationery Office, 1986). The aim of both of these schemes is to encourage environmentally sound farming practices.

The water levels on the moors are controlled to a large extent by pumps and sluices at the mouths of the main drains, and ditches are kept high throughout the summer to provide effective field boundaries and drinking water for stock. Maintaining a high water-table, cutting late for hay (July onwards) and excluding grazing cattle until midsummer, or allowing spring grazing only at very low stocking densities, are all important factors in allowing wading birds such as snipe (*Gallinago gallinago*) and redshank (*Tringa totanus*) to breed successfully (Green, 1986). These practices are characteristic of traditional farming in the area (Nature Conservancy Council, 1977) and, coupled with the customary use of little or no inorganic fertiliser, are also responsible for the great diversity of flowering plants in the meadows and ditches.

Aftermath growth in hay meadows is normally grazed at low stocking rates until October or November, either intermittently with dairy cows or continuously with beef cattle and young stock. Grazing with sheep is much less common. Low-intensity grazing at this time of year leaves a patchy vegetation structure in the autumn which is beneficial to overwintering birds (Green, 1986), but grazing will also influence the cycling of N (Jarvis et al., 1989a,b), resulting in more inorganic N available for leaching and denitrification compared with swards where the secondary growth is cut and harvested (Barraclough and Jarvis, 1989).

Both conservationists and agriculturalists have recognised the potential vulnerability of this environment to the effects of increasing soil fertility, but they have also been aware that there was little scientific information about the behaviour of inorganic fertilisers in wetland peat soils. This subject is of special significance in view of the potential damage that nitrate leaching could cause to the ecology of the ditches and watercourses through eutrophication (Stewart et al., 1982). Nitrous oxide (N_2O), one of the end-products of denitrification, is known to be damaging to the atmosphere both by ozone depletion and as a potent 'greenhouse' gas (Bouwman, 1990), so that the influence of fertiliser N application on denitrification rate is also of environmental significance.

In the experiment reported here a range of fertiliser N levels were applied to meadows managed by hay cutting and aftermath grazing; preliminary data for changes in agricultural output and botanical composition have been summarised elsewhere (Kirkham and Wilkins, 1989; Wilkins et al., 1989). Soil mineral N status was also monitored regularly over a 40 month period and denitrification rates were estimated during the autumn–winter of the latter 2 years.

This study was not intended to produce a complete balance sheet of N inputs and outputs, nor to characterise in detail the physical and chemical processes involved in the cycling of N in this particular context. So, although losses by ammonia volatilisation may have been significant while cattle were grazing the plots (Jarvis et al., 1989b) these were not measured, nor were inputs by atmospheric deposition. Instead, the main emphasis is placed on the autumn–winter period, when both rainfall and water-tables were at their highest and soil temperatures at their lowest. The potential for N leaching would therefore be greatest during this period, whilst frequent waterlogging and low soil temperatures would together tend to inhibit mineralisation and nitrification. Measuring the cumulative decline in soil mineral N between October and March each year therefore gave reasonable estimates of losses by leaching and denitrification. By estimating denitrification losses over the same period, leaching losses could be estimated by difference.

MATERIALS AND METHODS

The experiment is sited on 21 ha of hay meadows on Tadham Moor, in the Brue Valley near Wedmore in Somerset, in the southwest of England (51°12′N, 2°49′W). The soil consists of peat to a total depth of 125–160 cm over silty clay. Topsoil consists of 10–20 cm of black humified peat with a well-developed fine granular structure above a more weakly structured layer of humified peat to a depth of 40–70 cm. These upper layers are of raised moss origin, falling within the Turbary Moor, Altcar, Blackland and Adventurers series (Avery, 1980), with pH at 0–10 cm averaging about 5.7. The meadows have a long history of late hay cutting followed by aftermath grazing by cattle, with no underdrainage and no previous record of inorganic fertiliser usage.

Fertiliser application

Five levels of N, i.e. 0, 25, 50, 100 and 200 kg ha^{-1} year^{-1} (N-0–N-200) were applied each year from 1986 to 1989 to plots of between 0.6 and 1.1 ha, laid out in three randomised blocks, although the N-25 treatment was not included in any of the soil N assessments (see Table 1). Treatments were applied as granular ammonium nitrate (34.5% N), using a Bamlett tractor-mounted pneumatic distributor. Annual rates were split between two equal dressings, the first as soon as ground conditions allowed after mid-April and the second after the removal of a hay crop in July each year (August in 1988). Actual application dates were 14 May and 23 July 1986, 24 April and 27 July 1987, 18 April and 23 August 1988, and 3 May and 24 July 1989. On the first occasion each year the tractor was fitted with low ground pressure 'Terra Tires' to accommodate wet ground conditions. Phosphate and potash were applied in mid-season each year, in the form of triple superphosphate and muriate of potash. The rates, calculated on the basis of chemical analyses of hay swath samples, were sufficient to replace the amounts removed in hay. In order to minimise the effect of the experiment on the plant ecology of the meadows, no P or K was applied to control (N-0) plots. Averaged over 4 years, the mean amounts applied per year were equivalent to between 8.8 and 9.6 kg phosphorus ha^{-1} and 58.8–65.8 kg ha^{-1} potassium.

Sward management

Swards were cut for hay in July each year (August in 1988) and the aftermath growth was grazed with 12-month-old beef steers until mid- to late October. A continuous variable stocking regime allowed optimum utilisation of sward growth on each field, with stocking rates adjusted regularly to maintain a compressed sward height of 5.5–6.0 cm measured by a rising plate meter (Holmes, 1974).

Soil sampling

Table 1 gives details of the soil sampling programme. On each occasion a 25 mm diameter corer was used and sampling sites were chosen randomly within each plot.

Soil for mineral N content analysis was bulked to give a single sample for each depth on each plot, and these were crumbled by hand and mixed thoroughly before analysis. An incubation method, similar to that developed by Ryden et al. (1987), was used to determine denitrification rate. This used acetylene (C_2H_2) to inhibit the reduction of nitrous oxide (N_2O) during denitrification and measuring the amount of N_2O produced in a given time (Ryden et al., 1979). Soil was taken to a total depth of 30 cm at each sampling site, in three separate 10-cm cores. These cores were placed intact in 1000-ml fruit jars, together with the cores from two other sampling sites, to give nine 10-cm cores per jar. Each jar was sealed with a polyacetyl lid fitted with a rubber gasket and incorporating two septum seal stoppers. Acetylene (50 cm^3)

TABLE 1

Soil sampling details

	Soil mineral N	Denitrification
Sampling horizons (cm from soil surface)	0–10, 10–20, 20–30	0–30
No. of field samples per horizon from each plot	15–21 (1)[1]	12 (4)[1]
Sampling frequency	Monthly July–March 1987–1988 and 1989–1990[2], August–March 1988–1989 plus additional assessments on 11 November 1986, 19 March 1987, and 8 June 1989	1988–1989 and 1989–1990: Weekly mid-October–late December, then monthly until March[2]
Treatments sampled	N-0, N-100 and N-200 throughout, plus N-50 from August 1988 except 8 June 1989	N-0, N-100 and N-200 throughout

[1]Bracketed figures represent the number of samples per depth from each plot to which field samples were bulked for analysis.
[2]No assessment in February 1990 owing to flooding.

was injected into each jar on the day of sampling, maintaining atmospheric pressure within the jar by venting through a hypodermic needle inserted into the second septum during injection. Each jar was then fitted into a separate hole in the ground and incubated for 22–24 h. The exact incubation period and mean temperature were recorded for each batch of samples.

Sample analyses

At each mineral N sampling a subsample of 99.5–100.5 g was taken and analysed for mineral NH_4^+ and NO_3^- nitrogen by extraction in molar KCl solution (Whitehead, 1981). A second subsample was weighed fresh, dried overnight at 100°C and reweighed to obtain a measure of the moisture content of the soil. The extractant was passed through a segmented flow autoanalyser, combining a spectrophotometer to measure the ammonium content and a colorimeter to measure nitrate content after reduction of nitrate to nitrite on a cadmium column (Henriksen and Selmer Olsen, 1970). Any nitrite already present in the soil would therefore be included with nitrate in the analysis. However, this is unlikely to cause any significant inaccuracy, because nitrite usually occurs only transitorily and in small amounts in newly flooded soils as a primary breakdown product of nitrate in denitrification (Firestone, 1982).

After incubation of the denitrification samples, 2.5 cm^3 of air was removed from each jar and analysed for N_2O content on a Pye Unicam gas chromatograph, using a commercially available gas of known N_2O concentration in nitrogen (N_2) for calibration. The whole soil contents of each jar were then weighed before oven-drying for moisture determination.

Water-table, rainfall and temperature measurements

A total of 33 dipwells, each consisting of a 100 cm perforated plastic pipe 7.5 cm in diameter, were installed in late February 1986, sited in three parallel lines over the whole experimental area. This gave between one and three dipwells per plot, depending upon the size and shape of each plot. The depth of the water-table below ground level was measured at weekly intervals between 1 April and 14 November 1986 and since April 1987. A levels survey of the position of each of these dipwells in 1988 showed that ground level ranged from 2.13 to 2.49 m above Ordnance Datum.

Rainfall was measured continuously using a Meteorological Office Mark 2 tilting syphon gauge and weekly accumulated rainfall was also recorded in a simple bucket-type recorder.

Temperature recordings were made at 15-min intervals from April 1987 onwards in air (inside a Stevenson Screen), at the soil surface (under-grass) and at 10 cm depth in the soil using a Grant 'Squirrel' automatic meter/logger. Table 2 gives monthly average soil temperatures at 10 cm.

Analysis of data

Data for soilwater nitrate concentration were analysed for treatment and depth effects by analysis of variance, on the basis of a randomised block split-plot design, with N treatments classed as main plots. A separate analysis was performed for each date using GENSTAT 5 (Payne et al., 1987). Because the spatial relationship between depths was the same across all plots, and particularly because each depth horizon was sampled at every sample point, little weight should be placed on depth by treatment interactions. However, standard error bars representing error estimates for this interaction are given in Fig. 3 as a broad indication of the variation in the data at each assessment.

The total amounts per hectare of NH_4^+ and NO_3^- N in the top 30 cm of soil were calculated for each plot on each sampling date and these values, as well as total mineral N (i.e. $NH_4^+ + NO_3^-$) were subjected to analysis of variance for treatment effects. Polynomial contrasts (Payne et al., 1987) were fitted to the data to investigate linear, linear over quadratic, and additional cubic responses to levels of applied N. Data for the amount of N denitrified per hectare per day at each assessment in 1988–1989 and 1989–1990 were treated similarly.

The total amount of N lost from each plot during the autumn and winter (October–March) was calculated for each of the 3 years 1987–1988, 1988–1989 and 1989–1990. These figures were arrived at by accumulating the re-

TABLE 2

Monthly averages for daily minimum, maximum and mean soil temperatures recorded at a depth of 10 cm between April 1987 and March 1990

Year		January	February	March	April	May	June	July	August	September	October	November	December
1987	Min	-	-	-	9.6	10.6	15.2	16.4	15.5	11.4	7.4	6.7	6.0
	Max	-	-	-	12.3	14.7	16.9	17.7	17.1	18.9	14.9	9.9	9.1
	Mean	-	-	-	10.8	12.8	15.8	16.8	16.1	14.3	10.3	7.8	8.0
1988	Min	4.7	3.5	5.5	6.8	11.6	13.9	14.0	15.2	13.7	10.1	3.2	5.5
	Max	7.8	8.3	10.4	14.0	17.2	17.9	17.2	16.9	15.0	13.0	8.8	9.2
	Mean	6.2	5.5	7.4	10.0	14.1	15.6	15.5	16.0	14.3	11.5	5.9	7.6
1989	Min	5.6	5.6	6.7	6.8	13.3	14.4	16.8	14.7	13.2	7.6	7.2	5.0
	Max	7.5	7.2	8.7	9.1	16.4	17.0	20.2	19.4	16.7	15.4	8.8	6.3
	Mean	6.7	6.6	7.7	8.1	15.0	16.2	18.8	17.1	15.6	12.1	8.1	5.9
1990	Min	6.1	6.7	7.0	-	-	-	-	-	-	-	-	-
	Max	7.7	8.4	9.3	-	-	-	-	-	-	-	-	-
	Mean	6.9	7.6	9.3	-	-	-	-	-	-	-	-	-

sults of subtraction of each month's value from that of the previous month, with minimum results constrained at zero. This assumes that any increase in soil N from one month to the next is the result of a net gain in mineralisation over leaching and/or denitrification.

Total losses of N per hectare by denitrification between October and March in 1988–1989 and 1989–1990 were estimated. These were calculated first on a week-by-week basis, using the running mean rate of loss from two assessments, so that the rate for Week 1 was taken as the mean of Assessments 1 and 2, that for Week 2 from Assessments 2 and 3, and so on; similarly the amounts denitrified between late December and March each winter were calculated using running means of the monthly assessments during this period. Losses over the whole period were then totalled for each plot and tested for treatment effects by analysis of variance, with fitted polynomial contrasts. In addition, total N losses for the winter 1987–1988, 1988–1989 and 1989–1990 were tested for response to the levels of applied N both by linear and exponential regression, i.e. using the generalised formulae $y=a+bx$ and $y=a+b\times r^x$, respectively (the latter is equivalent to $y=a+b\times \mathrm{EXP}[-k(x)]$).

RESULTS

Changes in soil mineral N status

Patterns of soil N change in the top 30 cm of soil were broadly similar each year (Fig. 1). On plots receiving fertiliser, levels peaked soon after the mid-season application, returning to low levels by late winter. There were also seasonal fluctuations in nitrate (NO_3^-) N levels on control plots, with highest levels recorded in August or September each year. There was no clear evidence of any progressive build-up of extractable N with any of the treatments during the course of the experiment, although NH_4^+ N levels reached higher peaks on N-200 plots in 1989 than in either of the two preceding years.

For most of each year soil N was largely in the NO_3^- form, so that total mineral N patterns followed those of NO_3^- quite closely. Testing for linear response to applied N rates was generally much more effective in detecting treatment effects than straight analyses of variance. Figure 1 gives the probability levels for this response for each assessment date by star ratings. Ammonium N levels were only affected by applied N at these assessments in August 1987, November 1988, and September 1989. By contrast, NO_3^- and total extractable N levels showed significant linear responses at a majority of assessments, particularly when levels were high in summer and autumn. Differences among treatments diminished throughout the winter months and were negligible by March each year, although a significant linear relationship with applied N was still detectable for NO_3^- N on 27 March 1990.

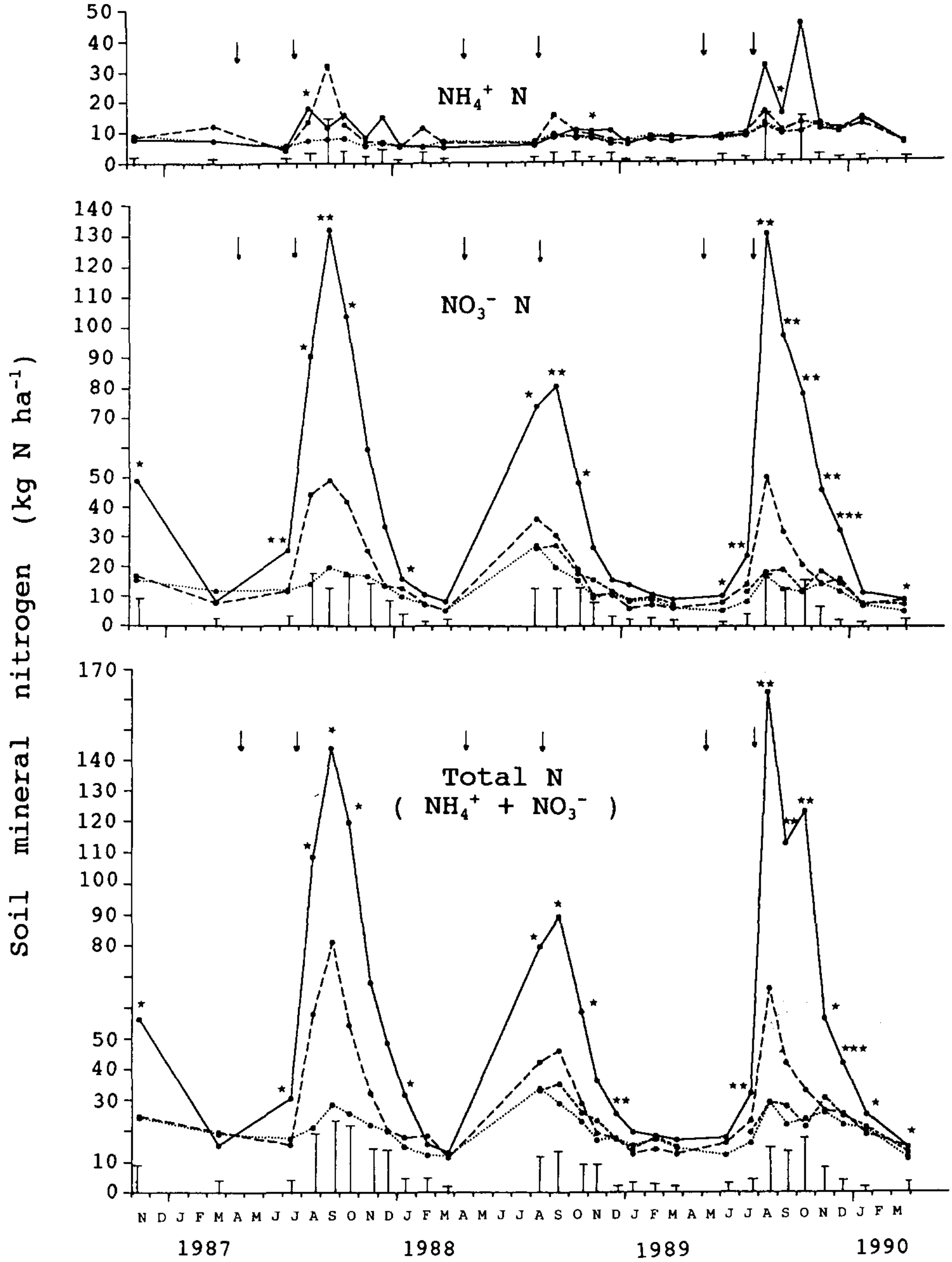

Fig. 1. Changes in soil mineral nitrogen in the top 30 cm of soil between 11 November 1986 and 27 March 1990. Dotted lines, no applied N; broken lines, 50 kg N ha^{-1} year^{-1}; larger broken lines, 100 kg N ha^{-1} year^{-1}; solid lines, 200 kg N ha^{-1} year^{-1}. Vertical bars are effective standard errors for treatment means. Asterisks denote significance levels of linear response to applied N: *$P<0.05$; **$P<0.01$; ***$P<0.001$. Arrows indicate timing of fertiliser application.

Soil N peaked at lower levels in 1988 than in either 1987 or 1989 (complete data were not available for 1986). Moreover, 1988 results differed from those of other years in that there were notable increases in soil N on all plots — particularly treated ones — between March and the August N application, despite an intervening hay cut. The amounts of N harvested in hay each year (Table 3) were calculated by multiplying the dry matter (DM) yield by the N concentration in the herbage. Because treatment differences in herbage N concentration were small, the amounts of N harvested in hay corresponded quite closely with the amounts of herbage DM harvested. Consequently, differences between treatments in the amount of N removed in herbage were smaller following the delayed harvest in 1988 compared with either 1987 or 1989 (DM response to applied N was almost certainly restricted in 1986 by low P and K availability, because no P or K had been applied by that stage in the experiment). However, the mid-season fertiliser application in 1988 appeared to have little effect on subsequent soil N levels, again in contrast with other years.

The above trends correspond quite closely with differing patterns of rainfall and water-table depth among the 3 years (Fig. 2). Summer rainfall was considerably higher in 1988 than in other years. The mean water-table depth approached the top 30 cm of soil (the sampling horizon) by mid-July, and much of the area was under water for 2–3 days in early October, although this occurred between dipwell readings and flooding was not, therefore, recorded. By contrast, the summer of 1989 was very dry and the water-table remained low until late October. Soil N levels were correspondingly high, but declined rapidly in response to wet weather from late October onwards. This high rainfall resulted in some flooding of the area in December and again in February 1990 (see Fig. 2).

TABLE 3

Amounts of herbage nitrogen (kg N ha^{-1}) removed in baled hay in 4 years 1986–1989

	1986	1987	1988	1989	Mean 1986–1989
N-0	52.0	42.4	68.4	43.9	51.7
N-25	43.4	49.0	60.8	48.6	50.4
N-50	57.2	55.0	80.9	61.8	63.9
N-100	53.2	57.8	74.8	58.8	61.2
N-200	65.7	85.9	88.6	81.6	80.5
SE	3.76	2.93	5.27	2.41	2.12
	*	***	*	***	**

Asterisks denote significance of treatment effects in analysis of variance *$P<0.05$, **$P<0.01$, ***$P<0.001$.

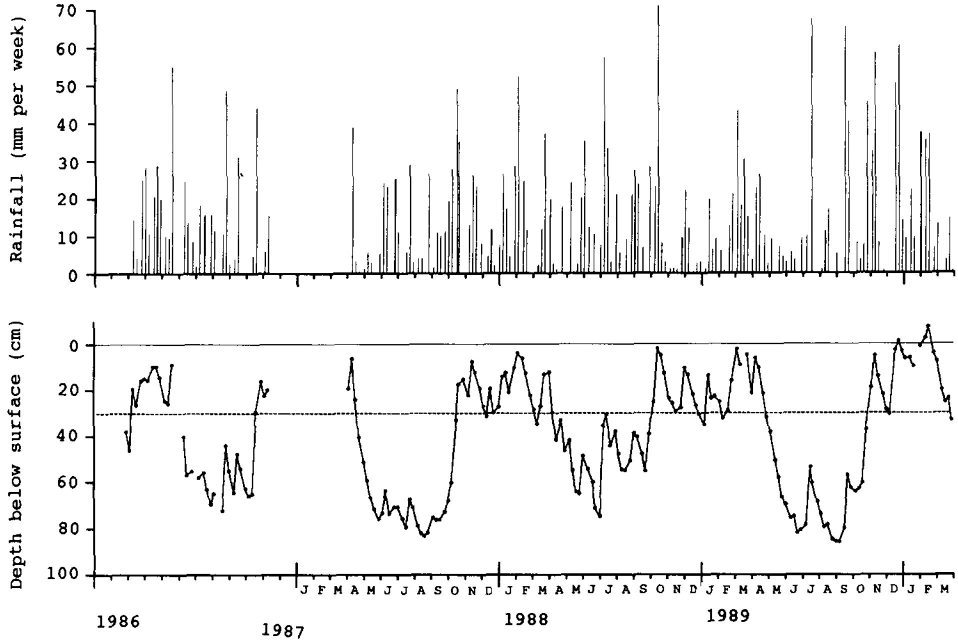

Fig. 2. Weekly rainfall totals (bar chart) and mean water-table depth in the 4 years 1986–1989 and part of 1990. The dotted line on the lower graph delineates the sampling horizon. Breaks in the water-table line indicate periods when no rainfall or dipwell readings were taken.

Nitrate concentrations in soil water

Figure 3 shows the nitrate concentrations in soil water in the three separate horizons from mid-October to March each year. However, the highest concentrations of NO_3^- N in the soilwater fraction were recorded at the first assessment following the mid-season N application each year (data not shown). This reached 167 mg l^{-1} in the upper 10 cm layer of N-200 soils in August 1987, nearly three times the concentration in the 10–20 cm layer. Once the water-table had reached the upper 30 cm horizon in October–November each year, differences between these layers diminished rapidly (Fig. 3). In both 1987–1988 and 1989–1990 the NO_3^- N concentration of ground water remained above 10 mg l^{-1} at all three depths beneath N-200 plots until January, with the sole exception of the 0–10 cm horizon in December 1989 (8.7 mg l^{-1}). By contrast, the NO_3^- N concentration beneath N-0 plots did not exceed 5.7 mg l^{-1} nitrate N (the current EC recommended limit for drinking water) in the 20–30 cm horizon after the September assessment in any year, and on N-100 plots remained at 5.1 mg l^{-1} or less between December and March for all 3 years.

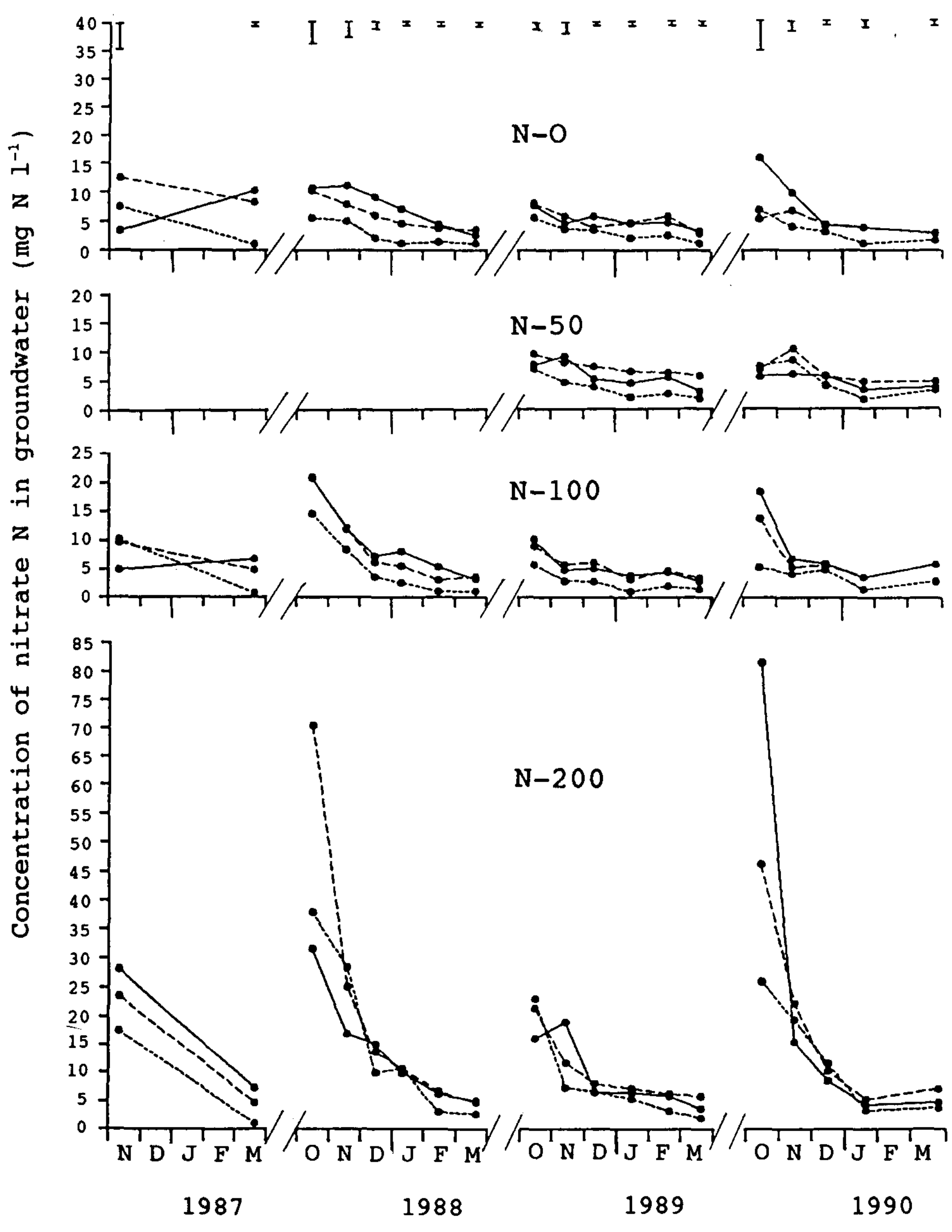

Fig. 3. Concentration of nitrate (mg l^{-1} NO_3^- N) in ground water at three depth horizons during the autumn and winter following contrasting fertiliser N applications. Solid lines, 0–10 cm; broken lines, 10–20 cm; dotted lines, 20–30 cm. Vertical bars are standard errors for depth × treatment means.

Denitrification

Denitrification fluctuated considerably from week to week, both in 1988–1989 and in 1989–1990, although the pattern of these fluctuations differed considerably among years (Fig. 4). In 1988–1989 rates tended to decline progressively from high levels in October, e.g. 328 g N day^{-1} at N-200, to less than 35 g day^{-1} for all treatments by the following March. By contrast, there was little denitrification occurring during October 1989, initially not exceeding 36 g N day^{-1} for any of the treatments assessed. Rates then rose to a peak on 13 November, reaching between 139 g day^{-1} (N-100) and 428 g day^{-1} (N-200). An ensuing trough over the next few weeks was followed by a second peak on all plots in mid-January, with rates declining progressively thereafter. Differences among treatments were significantly related to the amount of applied N at several of the assessments in both years. However, although there was usually a significant linear response on these occasions, this was invariably because of the influence of the N-200 treatment, with differences between N-0 and N-100 usually negligible.

Accumulated N losses October–March

Where no N was applied, accumulated losses of mineral N from the soil between October and March totalled 23.3 kg N ha^{-1} in 1987–1988, 12.6 kg

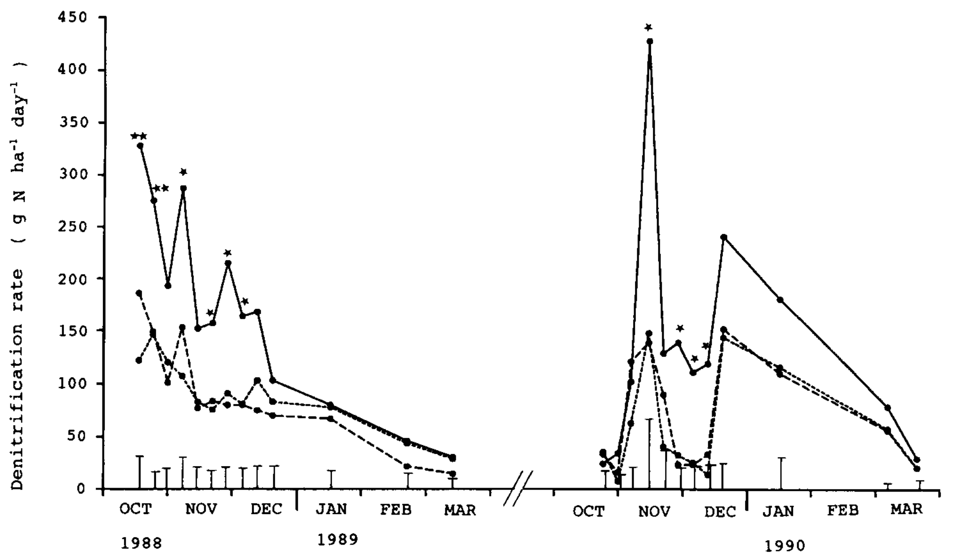

Fig. 4. Rates of denitrification (g N day^{-1}) in the top 30 cm of soil at individual assessments between October and March 1988–1989 and 1989–1990. Vertical bars are effective standard errors for treatment means. Asterisks denote the probability of a linear response to applied N: *$P<0.05$; **$P<0.01$. Dotted lines, no applied N; broken lines, 100 kg N ha^{-1} $year^{-1}$; solid lines, 200 kg N ha^{-1} $year^{-1}$.

ha^{-1} in 1988–1989 and 17.4 kg ha^{-1} in 1989–1990. In the latter 2 years denitrification accounted for losses of 11.3 and 13.3 kg N ha^{-1} respectively over the same period. On plots receiving fertiliser N, these losses varied considerably among years (Fig. 5). Although in 1987–1988 more than twice as much N was lost from N-100 soils compared with N-0, in 1989–1990 only the N-200 treatment stood out as significantly different from other treatments ($P<0.05$). In 1988–1989, losses from N-200 plots were little more than those from N-100 plots had been in the previous year. By contrast, accumulated denitrification losses for each treatment differed relatively little between 1988–1989 and 1989–1990, so that net losses of mineral N from N-200 plots, i.e.

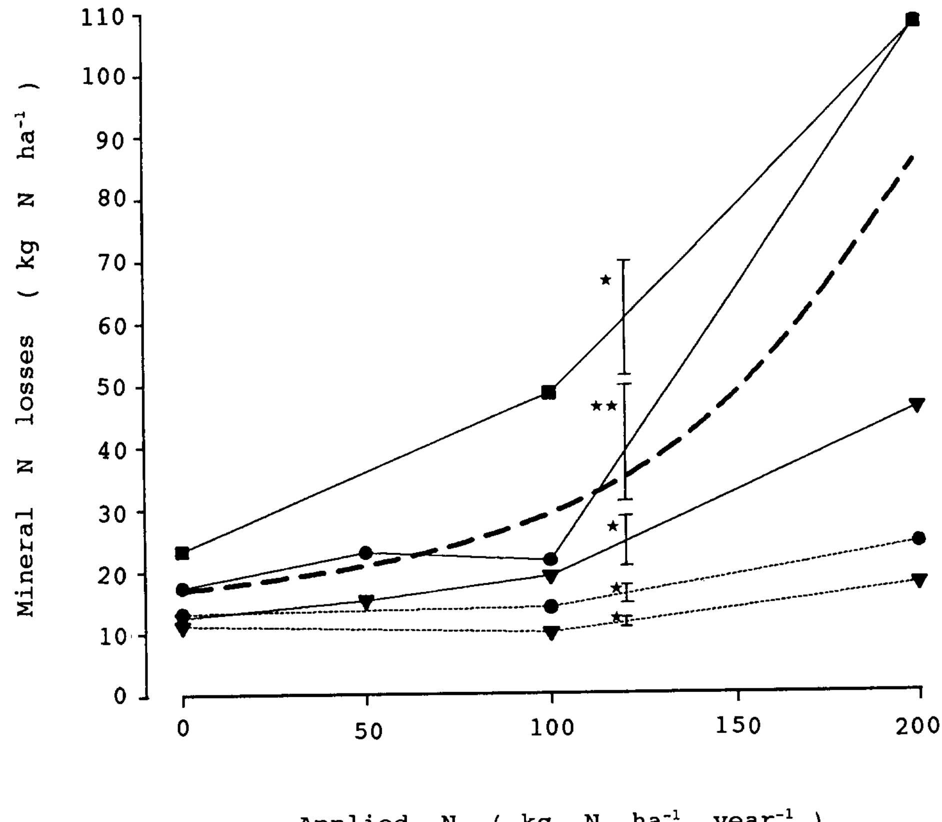

Fig. 5. Accumulated losses in total (ammonium plus nitrate) soil mineral N (kg ha^{-1}) between October and March in 1987–1988 (square symbols), 1988–1989 (triangles) and 1989–1990 (circles). Solid lines, total loss; dotted lines, loss by denitrification. Vertical bars are effective standard errors of treatment means; asterisks denote the probability level of a linear response to applied N: *$P<0.005$; **$P<0.01$. The thicker broken line represents an exponential curve fitted to treatment means for the 3 years (see text).

losses not accounted for by denitrification, were much greater in 1989–1990 than in 1988–1989, (84.0 kg N ha^{-1} compared with 28.2 kg ha^{-1}).

Figure 5 shows a fitted curve for total N loss averaged over the 3 years 1987–1988, 1988–1989 and 1989–1990. This used the treatment means for each year to derive the following formula: N loss (kg N ha^{-1}) $=13.9+3.2$ (1.0158^{N}), which explained 68.2% of the variance in the data. This equation implies that applying 100 kg N ha^{-1} $year^{-1}$ might be expected to increase total losses of soil N by an average of 12.2 kg N ha^{-1} over the autumn–winter period, and that applying twice that rate would incur additional losses of 58.2 kg N ha^{-1}. However, the above formula is clearly only valid within the range of N rates used in the model, because for rates above about 290 kg N ha^{-1} it predicts increases in N loss far in excess of the amounts applied.

Denitrification was not measured in 1987–1988. However, assuming that denitrification losses in that year were equivalent to the mean of the other 2 years for each treatment, then apparent leaching losses between October and March (i.e. total loss – denitrification) are estimated as 5 kg N ha^{-1} with no applied N, 17 kg N ha^{-1} $year^{-1}$ at 100 kg applied N ha^{-1} and 67 kg N ha^{-1} at 200 kg N ha^{-1} $year^{-1}$.

DISCUSSION

Periodic sampling of the soil for mineral N content is not a completely accurate method for measuring N loss, compared with more direct methods (e.g. Dowdell and Webster, 1980; Barraclough et al., 1984; Tyson, 1990) and denitrification rates often show wide temporal variation (Scholefield et al., 1990). However, care was taken in the approach to sampling and data analysis to minimise these inaccuracies, and the results give a good indication of the scale of losses that can result from the use of fertiliser N in wetland peat soils.

The main emphasis of this study was on estimating losses over the autumn–winter period. However, the summer peaks in soil N on N-100 and N-200 plots were notably lower in 1988 than in either the preceding or the following year, suggesting that significant losses occurred between August and mid-October in 1988. Moreover, the mid-season fertiliser application (on 23 August) had relatively little effect on amounts of soil N at the subsequent assessment, in marked contrast with other years. This was almost certainly the result of high summer rainfall and high water-table levels throughout the summer in 1988: a total of 57 mm of rain fell in the 22 days between fertiliser application in August and the next soil sampling, compared with 28 mm during the equivalent period in 1989 and only 8 mm in 1987. However, soil temperatures in August and September 1988 were no lower than usual and intermediate between those of 1987 and 1989. These factors would render soil N particularly vulnerable to both denitrification and leaching (Firestone, 1982;

Jarvis et al., 1989a). Losses of N measured between October 1988 and March 1989 will therefore probably represent a considerable underestimate of the total amounts lost during the whole year.

Ammonia N levels tended to be highest on all plots during the summer and early autumn. In N-0 soils the effect was small and attributable to increased mineralisation rates when soil temperatures were high, as well as to the direct supply of NH_4^+ N in the form of animal excreta during grazing (Jarvis et al., 1989a). Although NH_4^+ N peaked on N-100 and N-200 plots following fertiliser application, corresponding peaks in NO_3^- N were proportionately much greater. This implies that the NH_4^+ N applied (as ammonium nitrate) was rapidly nitrified to the NO_3^- form once in the soil. The ratios of NH_4^+/NO_3^- N were consequently much lower in these soils than in the silt loam and freely draining calcareous soils assessed by Jarvis et al. (1989a) at equivalent rates of N application, and the potential for losses by denitrification and leaching were correspondingly higher. In fact the peaks in denitrification rate recorded between October and March (328 g N day^{-1} in October 1988 and 428 g day^{-1} in November 1989) were high compared with rates recorded on mineral soils by Scholefield et al. (1990) and Ryden (1983), where equivalent peaks were recorded only immediately after fertiliser application to warm, wet soils. These results suggest high levels of biological activity in the peat soils at Tadham, leading to a high potential for both mobility and transformation of soil N. Nevertheless, denitrification could account for only a small proportion of the observed decline in soil N on N-200 plots, implying considerable leaching losses for this treatment.

It was not practicable within the scope of this study to measure the concentration of NO_3^- N in water percolating into the ditch system from individual plots, particularly as there were no underground drains present. Whilst dilution and denitrification of NO_3^- ions during their passage through the soil would have affected the concentration of NO_3^-, the mean concentration from N-200 plots would have almost certainly been above 11.4 mg l^{-1} nitrate N, the EC upper limit during most of the October–December period, easily exceeding the EC recommended limit of 5.7 mg l^{-1} for drinking water. These results make an interesting comparison with those from a drainage experiment in a heavy clay soil on sloping ground at North Wyke Research Station (Tyson, 1990). The run-off from undrained plots receiving 200 kg N ha^{-1} year^{-1} contained on average less than 10 p.p.m. NO_3^- N from late October onwards, with drainage water from equivalent drained plots remaining at above this level until the following January. The mean loss of N (over 7 years) from these soils was 18 kg N ha^{-1} year^{-1} and 58 kg N ha^{-1} year^{-1} for undrained and drained plots respectively, compared with an estimate of about 67 kg ha^{-1} year^{-1} for the N-200 treatment in this experiment.

Much lower leaching losses (1.3–18.1 kg N ha^{-1}) were shown at Letcombe Laboratory by Dowdell and Webster (1980). They measured leaching directly after applying fertiliser at 400 kg N ha^{-1} to monolith lysimeters containing a loamy sand soil sown with perennial ryegrass. However, in their work herbage was cut and removed six times per year and between 118 and 344 kg N ha^{-1} were harvested in herbage in the year of application. In grazed swards much of this herbage N would have been recycled onto the pasture and become available for leaching or denitrification; Jarvis et al. (1989b) found that N equivalent to 81% of that applied could be returned to the soil (at 210 kg N ha^{-1}), and of this proportion less than 5% is lost by ammonia volatilisation.

In a second experiment at Tadham Moor, run concurrently with the work reported in this paper, aftermath growth was cut instead of grazed (Kirkham and Wilkins, 1989). Herbage uptake of N in the aftermath phase was significantly improved by including higher rates of P and K than were used in the main experiment, reflecting a greater response in herbage production. Using these rates on the grazed plots might have resulted in a higher NH_4^+/NO_3^- ratio in soil mineral N by cycling a greater proportion of the available N through grazed herbage (Jarvis et al., 1989a), but the potential for reducing total soil mineral N is limited. Moreover, work by Campino (1982) suggests that amounts of mineral soil N might have been increased by additions of high rates of P and K together. Working with incubated silty loam hay meadow soil of 12.4% organic matter, Campino showed a significant P$\times$K interaction for increased N mineralisation, with the lowest rates used (equivalent to 87 kg P ha^{-1} and 166 kg K ha^{-1}) causing a significant 21% increase compared with control soils.

Barraclough and Jarvis (1989) introduced the concept of a break point for nitrate leaching, a level of fertiliser N application above which nitrate leaching increased markedly. Based largely on direct measurements of leaching and denitrification, they suggest break points of 150–200 kg N ha^{-1} for grazed swards and between 250 and 350 kg N ha^{-1} for cut grassland. The pattern of N loss in response to applied N varied considerably from year to year in the work reported here. Moreover, with a generalised curvilinear response, the identification of a single break point is both arbitrary and subjective. Nevertheless, it appears that the risk of substantial nitrate leaching begins somewhere between 50 and 100 kg N ha^{-1} year^{-1}, increasing markedly at rates above 100 kg ha^{-1}. This adds weight to the conclusion that these soils are at particular risk from nitrate leaching, especially as swards were managed by a combination of cutting and aftermath grazing and a break point might have been expected at somewhere between those quoted by Barraclough and Jarvis for grazed and cut swards.

Nevertheless, there does seem to be scope for improving N efficiency during the hay phase. Applying 50–100 kg N ha^{-1} in spring (N-100 and N-200) resulted in relatively little extra N being harvested in baled hay compared with the control. Much of the remainder will have been cycled through plants and returned in leaf material, either as plant tissue turned over during the accumulation of the late hay crop or by leaf shatter during hay making. In the above-mentioned small-scale experiment, high rates of P and K (75 kg P ha^{-1} and 200 kg K ha^{-1} $year^{-1}$) increased N uptake in the hay crop only marginally compared with taking an extra cut in late May–early June. A combination of these strategies could give the best utilisation of applied N in the hay crop. Unfortunately, cutting dates earlier than 1 July are unlikely to be acceptable in these meadows because of the deleterious effect they would have on the breeding success of wading birds, particularly snipe (Green, 1986).

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REFERENCES

Avery, B.W., 1980. Soil classification for England and Wales. Higher Categories. Soil Surv. Tech. Monogr., 14, 67 pp.

Barraclough, D. and Jarvis, S.C., 1989. The responsible management of nitrogen in grassland. In: Environmentally Responsible Grassland Management, British Grassland Society Winter Meeting, pp. 2.1–2.14.

Barraclough, D., Geens, E.L. and Maggs, J.M., 1984. Fate of fertiliser nitrogen applied to grassland. II. Nitrogen-15 leaching results. J. Soil Sci., 35: 191–199.

Bouwman, A.F., 1990. Exchange of greenhouse gases betweeen terrestrial ecosystems and the atmosphere. In: A.F. Bouwman (Editor), Soils and the Greenhouse Effect. Wiley, Chichester, pp. 62–126.

Campino, I., 1982. The effects of superphosphate and potassium fertilizer and salts on the nitrogen mineralization of incubated meadow soil. Fert. Res., 3: 325–336.

Dowdell, R.J. and Webster, C.P., 1980. A lysimeter study using nitrogen-15 on the uptake of fertiliser nitrogen by perennial ryegrass swards and losses by leaching. J. Soil Sci., 31: 65–75.

Firestone, M.K., 1982. Biological denitrification. In: F.J. Stevenson (Editor), Nitrogen in Agricultural Soils. Agron. Monogr., 22: 289–326.

Green, R.E., 1986. Breeding waders on the Somerset Moors; factors affecting their distribution and breeding success. A report to the Nature Conservancy Council. Royal Society for Protection of Birds, Sandy, CSD Report No. 625, 68 pp.

Henriksen, A. and Selmer Olsen, A.R., 1970. Automatic methods for determining nitrate and nitrite in water and soil extracts. Analyst, 95: 514–518.

Her Majesty's Stationery Office, 1986. The Environmentally Sensitive Areas (Somerset Levels and Moors), Designation Order 1986, Schedules 1 and 2 . Statutory Instrument No. 2252, 6 pp.

Holmes, C.W., 1975. The Massey Grassmeter. Dairy Farming Annu., 26: 26–30.

Jarvis, S.C., Macduff, J.H., Williams, J.R. and Hatch, D.J., 1989a. Balances of forms of mineral N in grazed grassland soils. Impact of N losses. In: R. Jarrige (Editor), Proceedings of the XVI International Grassland Congress, Nice, 1989, AFPF/INRA, Versailles, pp. 151–152.

Jarvis, S.C., Hatch, D.J. and Roberts, D.H., 1989b. The effects of grassland management on nitrogen losses from grazed swards through ammonia volatilization; the relationship to excretal N returns from cattle. J. Agric. Sci., 112: 205–216.

Kirkham, F.W. and Wilkins, R.J., 1989. Fertiliser effects in herb-rich hay meadows. In: Proc. 1st Research Meeting, British Grassland Society, Welsh Agricultural College, September 1988, Session II, Poster 12, British Grassland Society, Reading.

Nature Conservancy Council, 1977. The Somerset Wetlands Project. A consultation paper. The Nature Conservancy Council South West Region, Taunton, 25 pp.

Payne, R.W., Lane, P.W., Ainsley, A.E., Bicknell, K.E., Digby, P.G.N., Harding, S.A., Leech, P.K., Simpson, H.R., Todd, A.D., Verrier, P.J., White, R.P., Gower, J.C., Tunnicliffe Wilson, G. and Paterson, L.J., 1987. Genstat 5 Reference Manual. Clarendon Press, Oxford, 749 pp.

Ryden, J.C., 1983. Denitrification loss from grassland soil in the field receiving different rates of nitrogen as ammonium nitrate. J. Soil Sci., 4: 355–365.

Ryden, J.C., Lund, L.J., Letey, J. and Focht, D.D., 1979. Direct measurement of denitrification loss from soils. II. Development and application of field methods. Soil Sci. Soc. Am. J., 43: 110–118.

Ryden, J.C., Skinner, J.H. and Nixon, D.J., 1987. Soil core incubation system for the field measurement of denitrification using acetylene inhibition. Soil Biol. Biochem., 19 (6): 753–757.

Scholefield, D., Corre, W.J., Colbourne, P., Jarvis, S.C., Hawkins, J. and Klein, C.A.N., 1990. Denitrification in grazed grassland soils assessed using the core incubation technique with acetylene inhibition. In: N. Goborcik, V. Krajcovic and M. Zimkova (Editors), Proceedings of the 13th General Meeting of the European Grassland Federation, Banska Bystrica, Czechoslovakia, Vol. 2, pp. 8–12.

Stewart, W.D.P., Preston, T., Peterson, H.G. and Christofi, N., 1982. Nitrogen cycling in eutrophic fresh-waters. Philos. Trans. R. Soc. London, Ser. B, 296 (1082): 189–205.

Tyson, K.C., 1990. Report on AFRC–ADAS drainage experiment 1982–1989. AFRC Institute of Grassland and Environmental Research, Okehampton, 26 pp.

Whitehead, D.C., 1981. An improved chemical extraction method for predicting the supply of available soil nitrogen. J. Sci. Food Agric., 32: 359–365.

Wilkins, R.J., Hopkins, A., Kirkham, F.W., Sargent, C., Mountford, O., Dibb, C. and Gilbey, J., 1989. Effects of changes in fertiliser use on the composition and productivity of permanent grassland in relation to agricultural production and floristic diversity. In: R. Jarrige (Editor), Proc. XVI International Grassland Congress, Nice, 1989, France, AFPF/INRA, Versailles, pp. 101–102.