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Original article

Verifying the nitrification to immobilisation ratio (*N/I*) as a key determinant of potential nitrate loss in grassland and arable soils

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Abstract – The relative dominance of the competing pathways for ammonium, namely, the microbial processes of nitrification (*N*) and immobilisation (*I*), has been suggested as a major factor in controlling nitrogen losses from soils. In this paper we bring together data from four studies in arable and grassland soils to establish whether the ratio *N/I* is correlated with measured or modelled nitrate loss in drainage water. Individually, measurements of gross nitrogen transformations did not explain the variation in nitrate leaching across all sites. However, amounts of nitrate lost by leaching were well correlated with *N/I* for all sites. The relative importance of the pathways competing for mineralised nitrogen (expressed as *N/I*) was changed by management and this appears to be an important factor in controlling N loss in arable and grassland soils.

N losses / gross N transformations / ^{15}N isotope dilution / mineralisation / nitrification / immobilisation / *N/I* ratio

Résumé – Vérification que le rapport entre nitrification et immobilisation (*N/I*) est un facteur clé déterminant la perte potentielle de nitrate dans les sols cultivés et de prairies. La dominance relative des différents chemins concurrents pour l'ion ammonium, à savoir les processus microbiens de nitrification (*N*) et d'immobilisation (*I*), a été suggérée comme le facteur principal contrôlant les pertes d'azote à partir des sols. Dans ce papier, nous donnons en même temps des données de 4 études dans des sols labourés et de prairies pour établir si le rapport *N/I* est corrélé avec la perte de nitrates mesurée ou modélisée (dans l'eau de drainage). Des mesures individuelles des transformations globales de *N* n'expliquent pas la variation du lessivage de nitrates dans l'ensemble des sites. Cependant, les quantités de nitrates perdues par lessivage étaient bien corrélées avec *N/I* pour tous les sites. L'importance relative des chemins concurrents pour la minéralisation de l'azote (exprimé en *N/I*) a été changée par l'aménagement et cela apparaît être un important facteur dans le contrôle de la perte de N dans les sols labourés et de prairies.

pertes d'azote / transformations de l'azote totale / dilution de l'isotope ^{15}N / minéralisation / nitrification / immobilisation / ratio *N/I*

1. INTRODUCTION

The potential for soils to lose nitrogen (N) to the environment remains of pressing concern. Leaching of nitrate (NO_3^-) occurs widely, but is environmentally and economically undesirable [20]. The gaseous loss of N as nitrous oxide is also significant [1] and has important consequences for global warming.

The development of ^{15}N isotopic tracer techniques has enabled greater resolution and understanding of the processes that control N supply for plant uptake and N losses. The use of ^{15}N isotope dilution to measure the gross rates of soil N cycling processes was first described theoretically by Kirkham and Bartholomew [9]. However, only in the last 20 years has this theory been applied routinely to study gross N transformation rates in sediments and soils [e.g. 2, 3, 7, 12, 17].

These same concepts and techniques have been applied to obtain measurements of potential N loss and have been used for some years to assess the 'N-saturation' status of forest soils [21, 22]. In forest ecosystems and undisturbed grasslands,

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the N cycle was thought to be 'highly conservative', where high rates of mineralisation are balanced by high rates of immobilisation, resulting in minimal net N mineralisation or autotrophic nitrification [23]. Some nitrification occurs in soils, particularly natural ecosystems, through heterotrophic nitrification, which transforms organic N to NO_3^- directly. However, additions of N, e.g. through atmospheric deposition or fertilisation, can disrupt this balance and stimulate the process of autotrophic nitrification. If the system is unable to retain NO_3^- e.g. through plant uptake, increased losses of NO_3^- by leaching or denitrification may occur. Measurements of the rate of nitrification (*N*) using isotopic dilution, include both autotrophic and heterotrophic nitrification. However, in arable and grassland systems autotrophic nitrification has been shown to be dominant and *N* can be used to approximate the rate of ammonium (NH_4^+) consumption by autotrophic nitrification [15].

The relative dominance of the pathways of ammonium consumption via *N* or the immobilisation of N by the microbial biomass (*I*) can be expressed in the ratio *N/I*. Tietema and Wessel [21] suggested that forest soils with a high *N/I* have a greater potential to lose N from the system via leaching or denitrification than those with low *N/I*. This concept is also applicable to arable soils [14]. However, as yet this hypothesis remains untested against measurements of N loss in either arable or grassland systems. The aim of this paper was to establish whether the *N/I* index is correlated with measured, or modelled, NO_3^- loss in drainage water, by bringing together data from a number of studies in arable and grassland soils from the United Kingdom (UK) and New Zealand (NZ).

2. MATERIALS AND METHODS

Measurements of gross N transformations and measurements/estimates of NO_3^--N losses by leaching were made separately and independently in four studies at a number of sites in NZ and in the UK. Incubations of soil were carried out with both $^{15}NH_4^+$ and $^{15}NO_3^-$ to measure gross rates of mineralisation (*M*), *N* and *I* by ^{15}N isotopic pool dilution, either in the field or in laboratory studies (Tab. I). Details for each site are summarised briefly below giving references to the full published methods for each study as appropriate.

2.1. Grassland sites

2.1.1. Hamilton, NZ

2.1.1.1. Site

Four replicate paddocks of two sward management types (no fertiliser N, or 400 kg N·ha^{-1}·year^{-1} since 1993) at the Dexcel No. 2 dairy, Hamilton, NZ, were selected for measurement of gross N transformations in 1996 [11]. The permanent long-term pasture of predominantly perennial ryegrass (*Lolium perenne* L.) and white clover (*Trifolium repens* L.) was rotationally-grazed throughout the year by Friesian dairy cows stocked at 3.3 cows·ha^{-1}. In the fertilised farmlet, urea was applied in split applications of ca. 45 kg N·ha^{-1}, 1–4 days after each grazing, during all seasons except summer. Average total biological N_2 fixation in the unfertilised and fertilised farmlets was 174 and 40 kg N·ha^{-1}·year^{-1} respectively [11]. The soil was a free-draining silt loam of volcanic origin (Umbric Dystrochrept) with mean concentrations of organic carbon (OC) and total N in the soil (0–10 cm depth) of 4.66 and 0.51%, respectively. The long-term (75 year) mean annual rainfall for the site was 1200 mm [11].

2.1.1.2. Measurement of gross N transformations

In November 1996, two areas were selected within each of the four replicate paddocks of the two farmlets (giving eight plots in each farmlet). These were fenced to exclude grazing animals and the pasture was mown and the herbage discarded. In December 1996, the herbage was trimmed again and within each area, gross N transformation rates were measured in confined microplots (Tab. I). The full methodology used in this study is identical to that described in Ledgard

Table I. Description of methods used for measurements of gross N transformations in the 4 studies.

	Hamilton, NZ	Hillsborough, UK	North Wyke, UK	Arable sites, UK
Incubation type	Confined microplots in the field (76 mm diam. × 70 mm deep)	Soil cores, sieved and mixed in laboratory incubation (ca. 67 g soil)	Confined microplots in the field (100 mm diam. × 150 mm deep)	Soil cores, sieved and mixed in laboratory incubation (ca. 300 g soil)
$^{15}NH_4$ compound used	$(^{15}NH_4)_2SO_4$ (20 atom%)	$^{15}NH_4NO_3$ (20 atom%)	$(^{15}NH_4)_2SO_4$ (25 atom%)	$(^{15}NH_4)_2SO_4$ (60.9 atom%)
$^{15}NO_3$ compound used	$K^{15}NO_3$ (20 atom%)	$NH_4{}^{15}NO_3$ (20 atom%)	$K^{15}NO_3$ (20 atom%)	$K^{15}NO_3$ (61.2 atom%)
$^{15}NH_4$ and $^{15}NO_3$ application rate	1.26 g N·m^{-2} ca. 25 mg N·kg^{-1} (dry) soil	57 µmol N·g^{-1} 399 mg N·kg^{-1} (dry) soil	3 g N·m^{-2} ca. 20 mg N·kg^{-1} (dry) soil	1.5 mg N·kg^{-1} (dry) soil
Label application method	Injector	Surface applied and mixed	Injector	Surface applied and mixed
Sampling times	1 and 3 days	0 (10 mins) 1, 3, 7, 14, 21 days	1 and 4 days	1, 2, 7, and 14 days
Preparation of extracts for mass spectrometry	Distillation	Diffusion	Diffusion	Diffusion
^{15}N in microbial biomass	Measured	Not measured	Measured	Not measured
Method described in	[10]	[27]	[7]	[14]

et al. [10]. The data used in this synthesis are the average N transformation rates in each farmlet.

2.1.1.3. Determination of nitrate leaching

Nitrate leaching losses were determined using ceramic cup samplers (seven or eight per replicate paddock, 30 per farmlet) located at 1 m soil depth. Samples of solution were collected at ca. 2-weekly intervals and the NO_3^--N concentration was measured by flow injection analysis [11]. The total drainage volume was determined from lysimeters which were located next to the experimental area and comprised intact soil cores (0.4 m diameter, 1 m depth) which received 0 or 400 kg N·ha^{-1}·year^{-1} (four replicates) as urea at the same time as in the fertilised farmlet [11]. The amount of NO_3^--N leached was calculated from the temporal integral of NO_3^--N concentration and drainage volume. The data used in this synthesis are the flow weighted mean NO_3^--N concentration (mg·L^{-1}) and the average NO_3^--N leached (kg N·ha^{-1}) for each farmlet during the winter drainage period in 1996, the major annual drainage period closest to the measurements of gross N transformations.

2.1.2. Hillsborough, Northern Ireland, UK

2.1.2.1. Site

Five artificially drained perennial ryegrass swards, each 0.2 ha in area, were established in 1987 at the Agricultural Research Institute of Northern Ireland (NI) at Hillsborough, Co. Down, UK. The swards received 100, 200, 300, 400 or 500 kg N·ha^{-1}·year^{-1} applied as calcium ammonium nitrate in six equal dressings during the growing season each year. The swards were continuously grazed by beef steers from April to October each year to maintain a constant sward height of 70 mm [27]. The soil was a relatively free draining clay-loam (Typic Dystrochrept) with mean concentrations of OC and total N in the topsoil (0–75 mm) of 4.6% and 0.4%, respectively. The long-term (30 year) mean annual rainfall for the site was 884 mm.

2.1.2.2. Measurement of gross N transformations

Soil cores (3 cm diameter × 7.5 cm deep) were collected in January 1996 (during the main period of drain flow) and bulked from each of the grassland swards. The freshly collected soil was roughly sieved through a 8 mm sieve to remove large pieces of root and shoot material and gross N transformations were measured in a laboratory incubation (Tab. I). The full methods for the study are described by Watson and Mills [26]. Gross transformation rates at day 1, 3, 7, 14 and 21 were calculated from mean values of N concentrations and enrichments so that there was only one value per time interval, in contrast to the calculation methods used in the other studies [7, 10, 14]. The cumulative gross N transformations for each treatment were compared after fitting asymptotic curves to the relationship between the cumulative values and time [26]. Consequently, the data used in this synthesis are the cumulative gross N transformation rates for incubations for each sward.

2.1.2.3. Determination of nitrate leaching

The five swards were hydrologically isolated and artificially drained to V-notch weirs with weekly flow proportional monitoring and chemical analysis of drainage water [27]. Drainage water was routinely collected from the beginning of October to the end of February since drain-flow occurred during the autumn/winter period and generally commenced in October. Nitrate concentrations in leachate were determined using a TRAACS 800 continuous flow analyser [4]. Due to differences in drain efficiencies between plots, annual loads of NO_3^--N leached were estimated by multiplying the annual flow-weighted mean NO_3^--N concentration by the drainage volume (rainfall minus evapotranspiration for the drainage period) [27]. The data used in this synthesis are the flow weighted mean NO_3^--N concentration (mg·L^{-1}) and the average NO_3^--N leached (kg N·ha^{-1}) for each sward during drainage in 1995–96, the major annual drainage period closest to the measurements of gross N transformations.

2.1.3. North Wyke, SW England, UK

2.1.3.1. Site

In the summer of 1997, three farmlets were selected at the Rowden field drainage experimental site [24] on the IGER farm at North Wyke Research Station (Devon, UK) for the measurement of gross N transformation rates. The farmlets chosen had received N from N_2 fixation by clover (estimated to be 110 kg N·ha^{-1}·year^{-1}) or as 200 kg fertiliser N·ha^{-1}·year^{-1} or were unfertilised [24]. The farmlets had existed for at least 9 years and had been grazed by beef steers to maintain a sward height ca. 75 mm by adjusting the stocking rate (average 5 steers·ha^{-1}) during the grazing season. The main species in the unfertilised sward were Bent (*Agrostis* sp.), Vernal Grass (*Anthoxanthum odoratum* L.) and Yorkshire Fog (*Holcus lanatum* L.) and in the fertilised treatment perennial ryegrass was dominant. White clover (*Trifolium repens* L.) had been sown in the clover sward in 1988, which now contained approximately 10% clover with a mixture of perennial ryegrass and other grasses. The soil in all the farmlets was a poorly drained, silty clay loam (Typic Haplaquept). All swards were drained by a system of field and mole drains. The mean concentrations of OC and total N in the soil (0–15 cm depth) were 5.78 and 0.63%, respectively. The long-term (30 year) mean annual rainfall for the site was 1054 mm [24].

2.1.3.2. Measurement of gross N transformations

During 1997, three plots (30 m^2) were sited in each of the farmlets; animals were excluded from these plots during the grazing season and the swards were maintained in a 3-cut silage system. The fertilised plots received N in April and again in June and July, so that by the start of the measurement period (August) the fertilised plots had received 160 kg N·ha^{-1}. In August, gross N transformations were measured in confined microplots within each replicate area (Tab. I). The full methods and results for this study are described in Hatch et al. [7]. The data used in this synthesis are the average rates of N transformation processes in each farmlet.

2.1.3.3. Determination of nitrate leaching

Each farmlet was hydrologically isolated from its neighbours with drainage water directed through a weir [24]. Leaching losses were obtained directly from measurements made on samples collected daily from the outfall of the weirs throughout the winter period (1996–97) when the soil was at field capacity. The samples were analysed for NO_3^--N by continuous flow analysis and the annual loads of NO_3^--N leached calculated from weekly NO_3^--N concentration multiplied by the drainage volume (rainfall minus evapotranspiration for each drainage period). The data used in this synthesis are the flow weighted mean NO_3^--N concentration (mg·L^{-1}) and the average NO_3^--N leached (kg N·ha^{-1}) for each farmlet during the winter drainage period in 1996–97, the major annual drainage period closest to the measurements of gross N transformations.

2.2. Arable sites

2.2.1. Arable sites, England, UK

2.2.1.1. Sites

After harvest in 1998, composite samples of soil (at least 5 kg) were collected from the plough layer (0–30 cm) of 12 field trials, selected from 20 which had been established to study yield response to N fertiliser, covering the breadth of rotations and soil types in England (Tab. II). One site (14) had received some manure prior to 1995. At each site two treatments from the trials (comprised of 3 replicate plots) were sampled, one which was unfertilised (zero) and one at the farm rate of N application (farm), which sought to achieve the economic optimum N application rate and consequently varied according to the agronomist's recommendation at each site.

2.2.1.2. Measurement of gross N transformations

On return to the laboratory, soil samples were stored at 5 ºC for a maximum period of seven days before being sieved (< 6 mm). Gross N transformations were then measured in laboratory incubations (Tab. I). The full methods and results for this study are described in Murphy et al. [14]. The data used in this synthesis are the average rates of the N transformation processes for each treatment at each site.

2.2.1.3. Modelling of nitrate leaching

Agronomic information for at least three years (and up to 5 previous years) was used to compile set-up files for the model SUNDIAL v. 93-7 [19], which simulates the net magnitude of N cycling processes on a weekly time-step and has been widely evaluated in UK arable soils [19]. Weather data were obtained for each site from the closest meteorological station on a weekly basis over the same period. Leaching losses of N to below 1.5 m depth (kg N·ha^{-1}) between harvest in 1998 and the cessation of drainage in spring 1999 were estimated using the model for each treatment at each site. These data were then used as part of this synthesis.

2.3. Calculations and statistics

Experiments where both gross N transformations and NO_3^--N leaching were measured were identified from the literature and through personal contact. The scientists responsible for these studies (co-authors in this paper) then collated their data, particularly the *N/I* ratio and NO_3^--N leached (kg N·ha^{-1}), for all studies within one spreadsheet. The data analyses carried out separately for the individual studies were discussed and compared. Where data in individual studies are reported as being significantly different (using tests at the 95% confidence interval) this reflects statistical analyses, which are reported in full in the published reports for each study. Further statistical analysis was carried out on the collated data using GENSTAT 5, release 4.1. This particularly

Table II. Topsoil characteristics (0–30 cm) and cropping information for arable sites in England sampled at harvest 1998.

Site	Average annual rainfall, mm	Texture	Topsoil pH	Total N, %	Cropping: Previous crop	Cropping: Current crop	N rate, kg N·ha^{-1} (No. of applications)
4	603	Silt loam	5.2	0.14	Winter wheat	Winter wheat	264 (3)
5	665	Sandy loam	6.9	0.25	Field peas	Winter wheat	222 (3)
6	635	Clay loam	7.7	0.16	Winter wheat	Potatoes	240 (2)
9	658	Sandy clay loam	5.9	0.17	Spring OSR[1]	Winter wheat	122 (2)
11	617	Clay	7.8	0.17	Winter OSR	Winter wheat	200 (3)
12	617	Clay	7.4	0.16	Winter wheat	Winter wheat	223 (3)
13	589	Clay	7.8	0.16	Winter wheat	Winter barley	188 (2)
14	591	Sandy clay loam	7.9	0.14	Winter wheat	Winter OSR	238 (2)
16	534	Silt loam	7.9	0.07	Fallow (set aside)	Winter wheat	130 (2)
17	599	Clay	7.0	0.33	Winter beans	Winter wheat	160 (2)
18	768	Silty clay loam	6.9	0.12	Celery	Winter wheat	200 (2)
19	678	Sandy loam	6.3	0.06	Winter wheat	Potatoes	212 (2)

[1]OSR = oilseed rape.

included regression and curve fitting to investigate the relationships of the rates of gross N transformations (*M*, *N*, *I*) and *N/I* with leaching losses and flow weighted NO_3^--N concentrations across studies.

3. RESULTS AND DISCUSSION

3.1. Grassland sites

The NH_4^+ contents of the grassland soils were all ca. 4–6 mg NH_4^+-N·kg^{-1} soil. However, the ratio of NH_4^+:NO_3^- was ca. 10:1 in the soils, which were unfertilised and those under the mixed grass/clover swards, but was lower (usually = 2) in the fertilised soils. The lower NH_4^+:NO_3^- ratio in the fertilised soils suggests that fertiliser increased nitrifier activity. This was supported by the measured gross nitrification rates in the Hillsborough soils, where cumulative nitrification increased significantly with increasing previous N input ([26], Tab. III). The average daily nitrification rates during the incubation were 3.5, 4.9, 5.6, 9.4 and 10.0 mg N·kg^{-1}·d^{-1} for the Hillsborough soils which received 100, 200, 300, 400 and 500 kg N·ha^{-1}·year^{-1}, respectively. Rates of nitrification recorded in the volcanic NZ soils (Tab. IV) were at least as high as those measured in the most heavily fertilised Hillsborough soils. High rates of nitrification in the NZ soils have been attributed to the presence of allophanic clay [18], however, the mechanisms are not well understood. Where soils have high rates of nitrification, values of ammonium consumption (*N* + *I*) are often much higher than *M* reflecting the nitrification of NH_4^+ derived from sources other than mineralisation e.g. residual NH_4^+ fertiliser, fixed NH_4^+.

Measured rates of gross mineralisation (*M*) in the North Wyke and NZ soils ranged from 3.0 to 5.4 mg N·kg^{-1}·d^{-1} and, at both sites, maximum rates were recorded in the soils in which clover was grown without added N (Tab. IV). This may reflect the contribution of more readily mineralisable residues from clover to the soil organic matter. Daily *M* and *I* rates were variable in the Hillsborough soils and depended on the time interval selected for the calculations. There was no significant difference between soils, with a single modelled asymptotic curve explaining *M* under all fertiliser treatments; similar analysis showed that *I* was also not significantly different between treatments ([26], Tab. III). Different fertiliser management histories did not affect the gross mineralisation-immobilisation turnover. The equivalent daily rates of *M* and *I* expressed in mg N·kg^{-1}·d^{-1} averaged for Hillsborough soils were 6.1 and 3.0, respectively, which were similar to the daily rates given for the other grassland sites (Tab. IV). These mineralisation rates are within the range reported for other grassland systems (e.g. USA, ca. 5 [5]; NZ, 1.2–7.0 [29]; UK, 4.1–8.3 [8] and 1.5–4.0 mg N·kg^{-1}·day^{-1} [13]) and represent considerable fluxes (mg N·kg^{-1} soil·d^{-1}) of N through the soil inorganic N pool.

In the work reported here, there were no significant differences in *M* or *I* between the treatments at each site (Tabs. III, IV) and no clear correlation of these measurements with NO_3^--N leaching losses. There was a strong relationship between cumulative gross nitrification (kg N·ha^{-1}) over 21 days and NO_3^--N leached (kg N·ha^{-1}·year^{-1}) (r^2 = 0.95) under the range of fertiliser treatments in the Hillsborough soils. However, there was no correlation of *N* with measured NO_3^--N leached (kg N·ha^{-1}) for the other sites. In contrast, the index *N/I* was significantly correlated with total amounts of NO_3^--N

Table III. Cumulative gross mineralisation *(M)*, immobilisation *(I)* and nitrification *(N)* rates (kg N·ha^{-1}) after 21 days incubation of Hillsborough soils (Northern Ireland) having previously received inputs of 100, 200, 300, 400 or 500 kg N·ha^{-1}·year^{-1}. The calculated nitrification to immobilisation ratio (N/I) and the flow-weighted mean NO_3^--N concentration (mg NO_3^--N·L^{-1}) is also given.

Previous rate of N applied	*M*	*N*	*I*	*N/I*	Flow-weighted mean NO_3^--N concentration
100	115	44	82	0.54	4.6
200	95	84	56	1.50	5.7
300	101	107	62	1.73	11.8
400	97	166	65	2.55	19.3
500	90	182	51	3.56	24.1

Table IV. Gross mineralisation (*M*),nitrification (*N*) and immobilisation (I) rates (mg N·kg^{-1}·day^{-1}) for grassland plots receiving no N fertiliser (0N), fertiliser (+N) and fixed N from clover. Means of 6 replicates (North Wyke) and 8 replicates (NZ) (SE in brackets). The calculated nitrification to immobilisation ratio (*N/I*) and the flow-weighted mean NO_3^--N concentration (mg NO_3^--N·L^{-1}) are also given.

Sward	Site	*M*	*N*	*I*	*N/I*	Flow-weighted mean NO_3^--N concentration
Grass 0N	North Wyke	3.6 (0.55)	1.0 (0.06)	2.8 (0.48)	0.4	0.9
Grass/clover	North Wyke	5.4 (1.14)	1.2 (0.33)	1.7 (0.34)	0.7	2.0
Grass + N	North Wyke	4.1 (1.41)	1.6 (0.45)	2.8 (0. 1.5 (0.35)	1.1	11.7
Grass/clover	NZ	5.4 (1.01)	11.9 (2.17)	6.3 (1.18)	1.9	3.7
Grass/clover +N	NZ	3.0 (0.40)	13.0 (2.81)	3.1 (1.20)	4.2	15.8

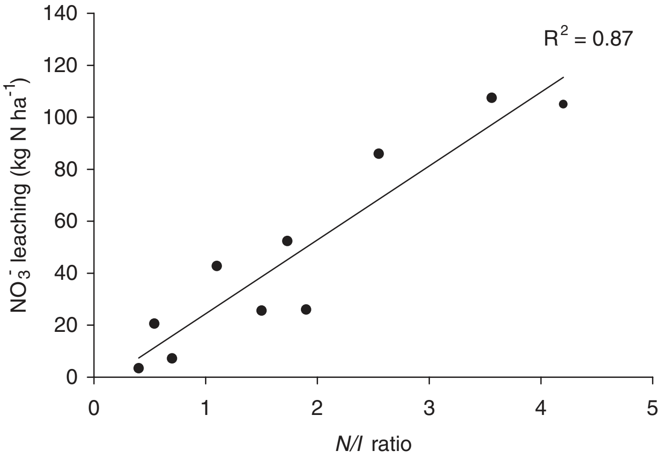

Figure 1. Correlation between the index *N/I* and measured nitrate (NO_3^-) leaching (kg N·ha^{-1}) in grassland systems.

leached at all sites (Fig. 1). The relative importance of the pathways competing for mineralised N (expressed as *N/I*) was changed by management and appears to be an important factor in controlling N loss in grassland. However, this index was not significantly correlated with annual flow-weighted mean NO_3^--N concentrations. For example the Hillsborough data show that a treatment with an *N/I* ratio below 1.7 had annual flow-weighted mean NO_3^--N concentrations below the EC limit (11.3 mg N·L^{-1}) for drinking water. In contrast, the grass/clover soil in NZ (with a *N/I* ratio of 1.9) had an annual flow-weighted mean of only 3.7 mg N·L^{-1} in the drainage, probably due to the differences in drainage volumes between the sites.

3.2. Arable sites

Following harvest in 1998, the topsoil at the arable sites contained a small NH_4^+-N pool (0.1–2 mg N·kg^{-1}) but a larger NO_3^--N pool (1.6–18.6 mg N·kg^{-1}). The ratio of NH_4^+:NO_3^- was ≤ 0.1 in all soils showing that nitrifier activity is high in these arable soils. There was significantly more NO_3^--N in the farm than the zero plots, but this amounted to an increase of only 3 kg N·ha^{-1}, on average. At these sites, normal farm practice did not leave significant pools of residual fertiliser in the topsoil at harvest.

Table V. Gross mineralisation rates (*M*, mg·kg^{-1}·day^{-1}) and gross nitrification rates (*N*, mg·kg^{-1}·day^{-1}) for arable plots either receiving no *N* fertiliser (Zero) or the standard farm application (Farm) at a range of sites in England measured after harvest 1998. Means of three replicates (SE in brackets). Missing values are indicated by n/a. The calculated nitrification to immobilisation ratio (*N/I*) is also given.

	Zero			Farm		
Site	*M*	*N*	*N/I*	*M*	*N*	*N/I*
4	1.1 (0.07)	1.2 (0.01)	1.7	0.9 (0.11)	1.0 (0.15)	1.6
5	0.7 (0.03)	0.9 (0.05)	1.5	0.4 (0.05)	0.9 (0.04)	6.2
6	0.4 (0.01)	0.8 (0.21)	3.5	0.4 (0.05)	0.8 (0.08)	6.2
9	1.1 (0.10)	0.8 (0.03)	0.9	1.1 (0.10)	0.8 (0.03)	1.0
11	1.1 (0.09)	1.4 (0.26)	1.4	0.9 (0.18)	n/a	n/a
12	0.7 (0.03)	1.1 (0.39)	1.5	1.1 (0.41)	1.2 (0.22)	1.3
13	1.0 (0.52)	0.9 (0.10)	0.9	0.6 (0.05)	1.1 (0.17)	2.0
14	0.8 (0.19)	1.0 (0.03)	1.3	0.5 (0.05)	0.8 (0.03)	1.8
16	0.6 (0.20)	0.7 (0.06)	1.2	0.1 (0.05)	0.01 (0.10)	0.2
17	0.5 (0.09)	1.8 (0.17)	0.1	1.1 (0.12)	n/a	n/a
18	0.5 (0.08)	0.7 (0.01)	0.5	0.5 (0.02)	0.9 (0.07)	0.1
19	1.1 (0.24)	1.0 (0.06)	0.3	1.3 (0.06)	0.7 (0.04)	0.2

The range of *M* measured in these arable soils (0.14–1.27 mg N·kg^{-1}·d^{-1}, Tab. V) is typical of previous measurements of *M* in arable soils [12, 13, 16, 25]. Rates of *N* ranged from 0.01–1.80 mg N·kg^{-1}·d^{-1} with no significant differences in *N* between sites, due to high variability within sites. This range is slightly broader than that previously measured for nitrification in other arable soils [28].

One arable soil (Site 12, Farm) had unusually large modelled leaching losses, which are likely to result from a failure of the model to simulate a previous pea crop in the rotation effectively, this was therefore excluded from an examination of the relationships between gross N transformation rates and modelled leaching losses. There were no significant relationships between *M* or *N* and modelled NO_3^--N leaching overwinter (kg N·ha^{-1}). However, there was a strong linear relationship between *N/I* and modelled NO_3^--N leaching in these arable soils (r^2 = 0.74; n = 21). When site 14, where manure had been applied in the rotation, was also excluded the

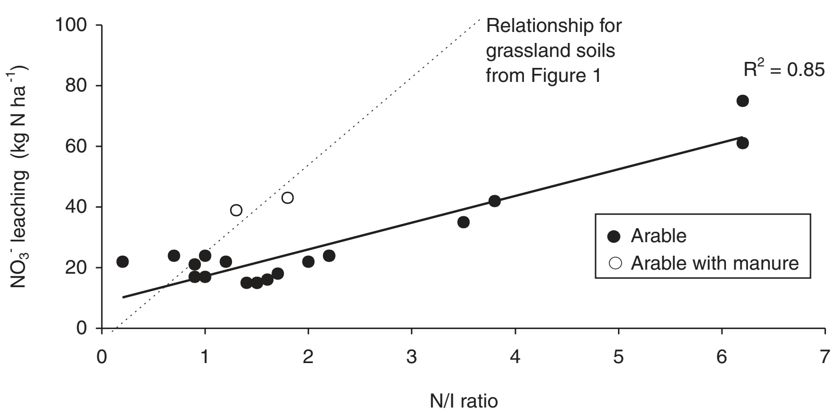

Figure 2. Correlation between the index *N/I* and modelled nitrate (NO_3^-) leaching (kg N·ha^{-1}) for arable systems.

strength of the relationship was increased ($r^2 = 0.85$; $n = 19$; Fig. 2). The site that had previously received manure seemed to fit the grassland relationship better than those for the other arable soils (Fig. 2). However, given that leaching losses were measured in the grassland soils and modelled in the arable soils, this may be merely coincidental. Further work is needed to investigate whether the relationships between measured NO_3^--N loss and *N/I* in arable and grassland soils are distinguishable. The *N/I* ratio was found to predict the likely NO_3^--N loss better than any of the rate measurements considered individually, which indicates that the relative importance of the pathways competing for mineralised N is a key determinant of overwinter NO_3^--N leaching in arable soils.

4. CONCLUSIONS

The synthesis of results from a number of studies has confirmed that the *relative* importance of the two pathways which compete for NH_4^+ (expressed as *N/I*) are important factors in controlling N loss through NO_3^--N leaching in both grassland and arable systems. The theoretical concept of the index *N/I* has been in use for almost a decade [21], but this work has confirmed its practical significance and value. Potentially *N/I* could be reduced through practical management options for fertilisers (e.g. using nitrification inhibitors) and residues (e.g. selection of crop types and/or grazing management to increase C:N ratio of residues). However, such changes in management may also have significant implications for other soil processes and should be considered carefully at field and farm scale.

Other work [6] has suggested that the productive and consumptive processes may not be equally affected by controlling factors such as temperature, moisture and soil structural changes. Consequently under certain soil conditions, the relationship between *N/I* and N leached is likely to differ from those found here. Additionally in most systems, mineralised N is only one component of the N that is leached. In soils with either significant pools of residual N fertiliser, or autumn additions of manure, it might be expected that more of the leached N will be derived from these pools than from mineralised N. Consequently the strength of the relationship between *N/I* and leaching losses of NO_3^--N in such situations might be weakened.

Calculated rates of *N* and *I* are usually associated with relatively large standard errors, due to the combination of measurement errors with relatively large multiplication factors [21]. While these can be propagated in the calculation of the resulting *N/I* ratio to allow its variability to be understood, it is most helpful to think of the *N/I* as an integrating value which gives a semi-quantitative measure of the capacity of a soil to produce NO_3^--N relative to its capacity to immobilise N. In addition, measurements of *N/I* are costly and labour-intensive and we do not envisage that this ratio will become a routine tool for the identification of sites with increased leaching potential. Nevertheless, there is clearly a need to try and identify the factors, which lead to the development of the range of *N/I* ratios that are observed in soils. These factors (if they can be identified) may, therefore, be more suited to routine measurements for monitoring leaching potential, or as inputs to models.

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