

What are the most important sinks for deposited nitrogen in UK moorland catchments?

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INTRODUCTION

Declining sulphur deposition in the UK will result in nitrate becoming the major acid anion in many acidified surface waters within the next decade, potentially preventing recovery from acidification (Curtis *et al.*, 2005a). Furthermore, there is increasing evidence that nitrogen (N) deposition may be affecting the nutrient status of remote upland waters that had previously been assumed to be phosphorus limited (Wolfe *et al.*, 2001; Maberly *et al.*, 2002; Sickman *et al.*, 2003). Modelling potential future increases in nitrate leaching is therefore of critical importance for the management of upland waters, which in the UK fall mainly within moorland catchments.

The steady-state mass-balance model FAB (First-order Acidity Balance model: Posch *et al.*, 1997; Henriksen & Posch, 2001) predicts future nitrate leaching by reducing the complex terrestrial and aquatic N cycles to the following long-term sinks for N:

1. In-lake retention of N (lake catchments only)
2. permanent removal of N in terrestrial biomass (grazing/harvesting)
3. Denitrification (terrestrial); return of N to the atmosphere in gaseous forms (e.g. N₂O)
4. Permanent immobilisation of N in soil organic matter.

These processes have been investigated in a series of government (DEFRA) funded studies in the UK. Most recently, N budgets were derived for four study catchments covering a gradient of N deposition and leaching, with an increasing proportion of N deposition leached as nitrate (Table I) according to modelled N deposition (1998-2001) and outflow fluxes derived from 2-weekly water chemistry and flow measurements (Curtis, 2003).

1. IN-LAKE RETENTION OF INORGANIC N

An earlier two year study of monthly water chemistry in the major inflow streams and outflows of 12 lakes (Fig. 1) across gradients of total N deposition and calcium concentration (as a measure of acid-sensitivity) found very little difference (<2% on average) in nitrate concentrations between inflows and outflows (Curtis *et al.*, 1998). Concentrations rather than fluxes are plotted, as flow data were not available. However, inspection of time series plots shows very little time lag between peaks in inflow streams and lake outflows, generally < 2-3 months. Hence short retention times may account for the very low in-lake retention of inorganic N.

2. N REMOVAL IN BIOMASS

In moorland catchments of the UK, the two potential routes for net N export in biomass are grazing (mainly sheep or deer at low density) and burning on grouse moors (INDITE, 1994). Both provide only very minor routes for N export in the catchments studied here.

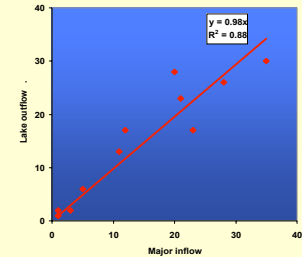


Figure 1: Comparison of mean nitrate concentration (µeq l⁻¹) in lake outflow and major inflow streams

4. N IMMOBILISATION IN CATCHMENT SOILS

The fate of N deposition onto the same four moorland catchments was assessed through the experimental additions of a stable isotope tracer, ¹⁵N labelled ¹⁵NH₄⁺, ¹⁵NO₃⁻ (Curtis *et al.*, 2005b).

The ¹⁵N tracer experiment showed that 65-99% of total inorganic N (TIN) inputs were retained by vegetation uptake or immobilisation in soils over one year of additions, leaving up to 35% of inputs potentially available for leaching, assuming that both denitrification and biomass removal represent very minor sinks for deposited N.

In the tracer study, N uptake by mosses and lichens appeared to exert the primary control on N leaching (Figs. 3-4). The implication is that leached nitrate is simply atmospheric deposition, transported in runoff, that is not retained by plants on the surface.

However, there are other possible mechanisms controlling nitrate leaching which could not be tested by the tracer work alone. For example, it has been hypothesised that the microbial production of nitrate may be linked to the C:N ratio of forest soil organic matter (Gundersen *et al.*, 1998). This hypothesis was tested for moorland soils at the four study catchments by Curtis *et al.* (2004), who concluded that while some sites may show a relationship between N deposition, C:N ratio and nitrate leaching it is also possible to get very high leaching from soils with a high C:N ratio, as at the River Etherow (Fig. 5). Further work by Evans *et al.* (in press) linked the susceptibility of catchments to nitrate leaching with the soil carbon pool.

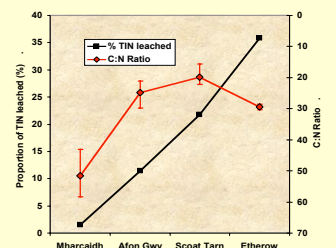


Figure 5: Relationship between inorganic N leaching and C:N ratio of surface organic layer averaged for all sampled soils (each with 3 replicates) in the 4 study catchments.

Table I: Description of the four experimental N budget catchments

SITE:	Allt a'Mharcaidh	Afon Gwy	Scoat Tarn	River Etherow
Region	Cairngorms,	Plymilton,	Lake District,	South Pennines,
	NE Scotland	Mid-Wales	NW England	NW England
Vegetation & land use	Wet & dry heath; nature reserve	Acid grassland; sheep grazing	Acid grassland; sheep grazing	<i>Calluna vulgaris</i> moor with burning
TIN depn. (kgN ha ⁻¹ yr ⁻¹)	6.4	15.7	22.5	30.7
Precipitation (mm)	1210	2258	2217	1272
% depn. leached as NO ₃ ⁻	1.6	11.5	21.7	35.8

3. DENITRIFICATION

The potential importance of denitrification in 4 moorland catchments (Table I) was assessed through a combination of in-situ monitoring and laboratory-based soil incubations. Field rates of denitrification were found to be very low (Table II), with a maximum flux in unamended soils (mean ± SD for 3 replicated plots per soil) of 0.24 kgN ha⁻¹ yr⁻¹ at the River Etherow. The maximum flux from a single plot peaked at 3.4 kgN ha⁻¹ yr⁻¹ (Fig. 2) but the annual mean value was only 0.68 kgN ha⁻¹ yr⁻¹ due to zero fluxes for much of the year. In field plots with added N mean fluxes were still <0.5 kgN ha⁻¹ yr⁻¹.

Laboratory incubations of soil cores from the same plots at 5°C and 15°C with and without added N showed that all soils were capable of large denitrification fluxes under optimal conditions of temperature, wetness and N availability but C limitation rapidly occurs. Such optimal conditions rarely occur in the field, hence the low annual mean fluxes observed in situ. Denitrification is therefore a very minor sink for deposited N at present and its importance is unlikely to increase greatly due to the very transient nature of optimal conditions that facilitate larger fluxes.

Table II: Annual mean soil temperature, soilwater chemistry & N₂O fluxes in field plots (kgN ha⁻¹ yr⁻¹)

Site	Soil code	Soil type	Temp. (°C)	pH	Conc. (µeq l ⁻¹) NH ₄ ⁺	NO ₃ ⁻	N ₂ O flux (kgN ha ⁻¹ yr ⁻¹)	Untreated	+NH ₄ ⁺ , NO ₃ ⁻
Allt a' Mharcaidh	M1	Peaty ranker	4.4	3.4	1.4	0.0	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00
	M2	Valley peat	5.5	3.7	1.4	0.0	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00
	M3	Peaty podsol	5.8	4.0	1.4	0.0	0.03 ± 0.05	0.03 ± 0.05	0.49 ± 0.84
	M4	Shallow peat	6.2	3.9	2.9	0.0	0.02 ± 0.03	0.02 ± 0.03	0.34 ± 0.38
Afon Gwy	G1	Hilltop peat	7.3	3.4	5.0	0.0	0.02 ± 0.03	0.05 ± 0.04	0.05 ± 0.04
	G2	Peaty gley	-	3.6	2.9	0.0	0.05 ± 0.05	0.05 ± 0.05	0.05 ± 0.05
	G3	Podsol	8.4	4.1	2.1	0.0	0.01 ± 0.02	0.01 ± 0.02	0.01 ± 0.02
	G4	Valley peat	8.3	3.8	3.6	0.0	0.12 ± 0.21	0.12 ± 0.21	0.12 ± 0.21
Scoat Tarn	S1	Podsol	6.5	3.9	2.1	8.6	0.00 ± 0.00	0.00 ± 0.00	0.00 ± 0.00
	S2	Peaty gley	6.6	3.8	2.1	10.7	0.01 ± 0.02	0.01 ± 0.02	0.01 ± 0.02
	S3	Deep peat	6.9	3.9	2.1	2.9	0.05 ± 0.05	0.05 ± 0.05	0.05 ± 0.05
River Etherow	E1	Deep peat (burnt <i>Calluna</i>)	7.2	3.0	20.0	92.1	0.24 ± 0.38	0.08 ± 0.08	0.08 ± 0.08
	E2	Deep peat (unburnt)	7.1	3.3	190.0	2.9	0.02 ± 0.03	0.00 ± 0.00	0.00 ± 0.00

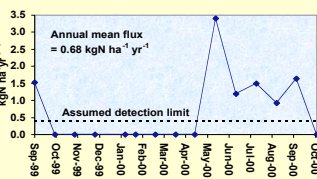


Figure 2: Monthly denitrification flux (N₂O: kgN ha⁻¹ yr⁻¹) from plot E1D1 at the River Etherow

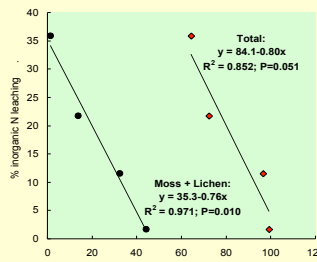


Figure 3: Relationship between % inorganic N leaching, total recovery of ¹⁵N inputs and recovery in moss + lichen. ♦ = total for all vegetation and surface organic matter compartments, • = moss + lichen. Points represent the mean value of 2-4 soils each with 3 replicates. Source: Curtis *et al.* (2005b)

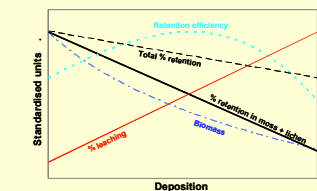


Figure 4: Schematic diagram of change in moss + lichen properties with increasing deposition. Source: Curtis *et al.* (2005b)

IDENTIFYING NITRATE SOURCES

Since the first three N sinks considered are all minor, the key to modelling nitrate leaching lies in the ability of catchment plants and soils to retain and immobilise deposited N.

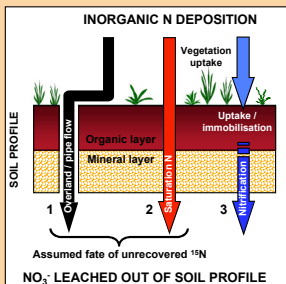


Figure 6: Schematic diagram of 3 potential nitrate leaching pathways. Source: Curtis *et al.* (2005b)

i.e. in the balance between abiotic (hydrological) and biotic processes. These may be summarised in three potential mechanisms (Figure 6). Although each is described separately, they are inevitably linked. Mechanism 1 may be considered as "hydrological nitrate" determined primarily by the physical characteristics of soil and vegetation. Mechanism 2, termed here "saturation nitrate", is present in excess of biological demand, or at least transported through the soil-vegetation system faster than it can be biologically utilised. The ¹⁵N tracer work showed that these two mechanisms could jointly account for observed nitrate leaching. The third mechanism, nitrification, may be controlled by soil C:N ratio but may not be important at all sites (see Fig. 5).

While the above studies, plus work by Evans *et al.* (2004) have provided circumstantial evidence for the atmospheric source of leached nitrate, current studies are using dual isotope methods (cf. Durka *et al.*, 1994; Spoelstra *et al.*, 2001; Willard *et al.*, 2001) to directly determine whether leached nitrate is atmospheric or produced microbially within soils.

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