

Nitrogen Loss from a Soil Restored after Surface Mining

R. Davies,* R. Hodgkinson, A. Younger, and R. Chapman

ABSTRACT

Anaerobic conditions develop below about 1-m depth in soil stockpiles and this can lead to an accumulation of ammonium (NH_4^+) and transformations of a normally relatively inert organic-N pool within the soil. After reinstatement of the soil from the stockpile, the NH_4^+ may rapidly be transformed to nitrate (NO_3^-) and lost from the soil with the labile organic-N. These losses may lead to a pollution risk in addition to the loss of a resource. The magnitude of N losses were measured from a soil that had been reinstated for agricultural use after being stored in a stockpile for 12 yr. Nitrogen movement in the soil profile and losses into the water-course were monitored for 2 yr beginning immediately after reinstatement. Relatively large movements of N were detected within the soil profile and large losses to the atmosphere and/or water-courses were estimated. Over the 2-yr monitoring period, 2449 kg ha⁻¹ of N was lost from the soil profile; 90% of this was not accounted for either in the soil or in the runoff or drainage water as mineral-N and was presumed to have entered either the atmosphere or aquatic environment as organic-N. To aid remedial measures, the proportion entering the atmosphere as nitrogen and nitrous oxides or the aquatic environment as organic-N, needs further investigation.

STOCKPILING OF SOIL in mounds during mineral extraction has been shown to affect the biological, chemical, and physical properties of the soil (Hunter and Currie, 1956; Barkworth and Bateson, 1964; Harris et al., 1989; Johnson et al., 1991). At depths below about 1 m in the mound, the numbers of anaerobic bacteria increase whereas those of aerobic bacteria decrease (Harris et al., 1989). Inhibition of nitrification because of poor aeration within the mound can cause an accumulation of NH_4^+ in the anaerobic zones. Once the soil is removed from the mound and reinstated, the aerobic microbial population rapidly reestablishes, usually at higher than normal levels (Williamson and Johnson, 1991), and nitrification recommences at greater than normal rates. If high levels of NH_4^+ are present in a reinstated soil, the amount of NO_3^- generated is likely to be much greater than normal and consequently there is a high potential for N loss to the environment via leaching and/or denitrification (Johnson and Williamson, 1994). Nitrate leached to water courses is not only a threat to aquatic environments and drinking water supplies (Addiscott et al., 1991), but if N is lost from soil in the form of gaseous nitrous or nitrogen oxides, this will contribute to the degradation of the ozone (Isermann, 1994). Toxicity of NH_4^+ to fish has been shown to occur at around 10 mg

L⁻¹ (Vinogradov, 1987) and toxicity of NO_3^- to occur at around 200 mg L⁻¹ (Tomasso and Carmichael, 1986). Apart from any pollution potential, a loss of N is also the loss of a resource.

This study investigated the magnitude of N losses occurring immediately after reinstatement of a soil that had been stored for 12 yrs during coal opencast mining. The organic N and mineral N content of the soil, and the N content of the drainage water and surface runoff were measured over a 2-yr period following reinstatement. A study on the same site will investigate the effect of cropping management on the N balance in the reinstated soil.

MATERIALS AND METHODS

Site and Soils

The site was located on the southern end of the Butterwell opencast coal site in Northumberland UK (Grid. Ref. NZ201888). Four hydrologically isolated plots each of 0.6 ha were established on soil that had been reinstated during August 1991. The plots were isolated by a perimeter drain installed at 0.9-m depth using a continuous chain trencher in a trench backfilled with gravel up to ground level. Due to the compact nature of the soil/overburden material found on restored opencast coal sites, water movements below 0.9 m are minimal (Davies et al., 1992). Normal practice, as part of the after care of restored land would be to install underdrainage after the first winter following reinstatement. In this instance, for experimental purposes, drainage was installed immediately after reinstatement. The soil before disturbance was a clay loam stagnogley of the Dunkswick series that exists extensively in the region. The soils are recorded as wetness class III in the undisturbed state and class IV/V in the restored state (Jarvis et al., 1984).

The soil profile that comprised two horizons overlying overburden material was a direct result of the reinstatement procedure (Fig. 1). Topsoil and subsoil had been stripped from the original site, and stored separately in mounds up to 5 m in height for 12 yr during the coal extraction period. All soil handling was carried out by box-scraper earth movers. Reinstatement consisted of replacing the overburden and levelling, replacing approximately 1 m (if available) of subsoil and finally replacing about 0.25 m of topsoil. Soil layers were evenly replaced to the final contour and generally of uniform depth. Very little or no mixing of topsoil with subsoil occurred during soil movements, however no effort was made to differentiate between soil stored at different depths within a mound. After reinstatement, the drainage system was installed (Fig. 1). The site was subsoiled and disced before perennial ryegrass (*Lolium perenne* L.) was sown in September 1991. In October 1992, all plots were ploughed and different crop management regimes were established and subsequently monitored. Details of this

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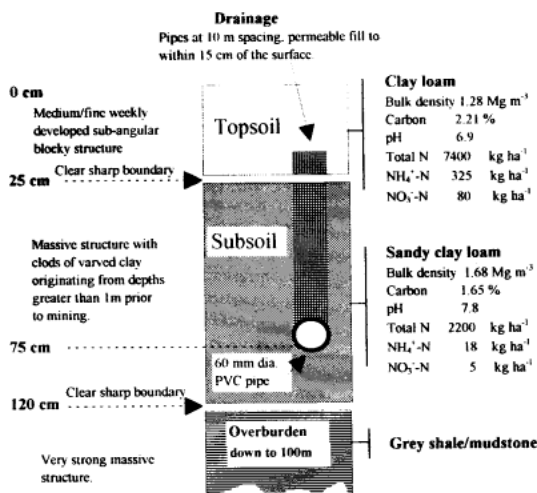


Fig. 1. Restored soil profile at Butterwell.

work will be reported elsewhere as during this study only the mean of the four plots is used.

Hydrological Monitoring

The drainage pipes and surface collectors from each plot were connected by sealed plastic main drains to a chamber housing the discharge monitoring equipment. Each discharge component was continuously (reading every 5 min) monitored using 1/2 90° V notch weirs (Talman, 1979). Data were recorded both autographically on charts by a float linked to a pen (Talman, 1983) and electronically on a Campbell CR10 data-logger, by use of a rotary potentiometer linker to the float. Rainfall was continuously monitored using a tipping bucket raingauge with a 0.2-mm graduation. Data were collected automatically by a radiotelemetry link each night.

Soil Sampling, Analysis, and Monitoring

Soil samples were taken using an auger at depths down to 0.75 m from October 1991 to October 1993. In the first year, 25 samples were taken from 0 to 0.25 m, 0.25 to 0.5 m and 0.5 to 0.75 m depths from each plot in an equally spaced grid pattern. In the second year 16 samples were taken from each depth. Samples were taken in October, March, and May or June of each hydrological year. Soil analysis was carried out using methods described fully by MAFF (1981). Levels of NH₄⁺ and NO₃⁻ were determined by leaching with potassium chloride. Total N was determined by Kjeldahl digestion with sulfuric acid, pH was determined in water and C content was determined by loss on ignition.

Soil moisture and temperature were monitored on a similar, restored nearby site up until October 1992 and from October 1992 to October 1993 they were monitored on the study site. Soil moisture content was determined every 2 wks at depths to 1 m using a neutron moisture meter. Soil temperature was monitored at 10 and 25 cm depths using a probe with a Campbell Scientific data logger.

Water Sampling and Analysis

Water samples were collected using EPIC automatic liquid samplers controlled by a program on the data logger. A com-

puter program on the data logger continuously monitored the flow and used an algorithm to target the samples on a flow proportional basis. This ensured that the maximum numbers of samples were taken when the greatest changes in flow rate were taking place. The sample volume collected was 200 mL from which a 20-mL subsample was taken, not filtered and frozen within 15 min, for later analysis. The subsamples were stored frozen until they were defrosted at 27°C before immediate analysis.

All analyses were carried out using an Alpkem autoanalyzer. Nitrate-N was determined colorimetrically after reduction to nitrite-N and diazotization with sulfanilamide (American Public Health Association, 1976). Ammonium-N was determined colorimetrically following reaction with alkaline hypochlorite and phenol to form indophenol blue (Patton and Crouch, 1977).

RESULTS AND DISCUSSION

Initially an account is given of the soil conditions at the start of the monitoring period immediately after reinstatement. The results for hydrology, water quality, and soil conditions are then presented and discussed within the periods of autumn/winter and then spring/summer for each of the 1991 to 1992 and 1992 to 1993 hydrologic years.

Soil Conditions Immediately after Reinstatement

Soil samples taken at the start of the experimental period in October 1991, 8 wks after the soil was replaced, indicated that more than 325 kg ha⁻¹ of the topsoil total-N was in the form of NH₄⁺ (Fig. 2a) compared to the more normal 10 to 20 kg ha⁻¹ at this time of the year in a similar undisturbed soil, (Restoration of Land after Opencast Mining, 1984, Report 2, NCBOE Northern Research Project, 1984, unpublished data) indicating that nitrification had been inhibited during soil stockpiling. The relatively low rainfall in the 1990/1991 hydrologic year (576 mm compared to the usual 670 mm: Smith and Trafford, 1976) and dry soil conditions prior to and around the time of reinstatement, may have meant that little transformation of soil N could have taken place between the times of removal of soil from the stockpile and sampling. Topsoil NO₃⁻-N content (Fig. 2b) was also relatively high at 80 kg ha⁻¹, compared to that previously found in a similar undisturbed soil in the region (10–20 kg ha⁻¹). The total amount of mineral and organic N in the topsoil (Fig. 2c) was initially similar to that previously found in an undisturbed soil of the same series in the region (7500 kg N ha⁻¹, Restoration of Land After Opencast Mining, 1984, Report 2, NCBOE Northern Research Project, 1984, unpublished data). Up until the time of soil restoration, N losses should have been minimal as during storage the mounds were not uncovered and leaching of accumulated NH₄⁺ would have been minimal.

Subsoil NH₄-N and NO₃-N contents were less than those in the topsoil and similar to those in an undisturbed subsoil at this time of the year (<10 kg ha, Restoration of Land after Opencast Mining, 1984, Report 2, NCBOE Northern Research Project, 1984, unpublished data). The pH, especially of the subsoil, had increased during stockpiling from the usual value of approximately 7 to

7.8, this may have been a consequence of the accumulation of NH_4^+ (Williamson and Johnson, 1991).

Hydrology, Soil Nitrogen, and Water Quality October 1991 to October 1992

Rainfall during the hydrologic year (1 Aug.–31 July 1991/1992) was below average, with a total annual rainfall of 497 mm being recorded, compared to the long-term average for the area of 670 mm (Smith and Trafford, 1976). The soil was nearly air dry (volumetric moisture content below 10%) prior to reinstatement in 1991 and, as a result of the low rainfall field capacity was not reached during the winter of 1991 and 1992 (as indicated from neutron probe soil moisture monitoring at a nearby site). As a consequence, the total drainage recorded during winter 1991 and 1992 was low at 120.1 mm and very little surface runoff occurred. The drainage path through the soil profile found to occur on a similar restored opencast site in the region (Bragg et al., 1984) and which very likely occurred on this site is dominated by flow in the topsoil or in the zone above subsoiling depth. The compacted subsoil severely retards water penetration and as a consequence water flows along the top of the subsoil (interflow) until it enters the permeable fill over the drains. This may occur even after subsoiling operations. Frequently (especially in dry years) field capacity of the subsoil is not reached (Davies et al., 1992) and the lower zone of the topsoil becomes saturated.

Autumn and Winter 1991/1992

The accumulation of NH_4^+ was rapidly removed over the period October 1991 to February 1992 (Fig. 2a) and the proportion of the total N present as NH_4^+ in the soil decreased from 4.4 to 0.9% whereas the proportion present as NO_3^- increased from 1.1 to 2.7% (Fig. 2b) some of which entered the drainage system. Johnson and Williamson (1994), found approximately the same pattern of NH_4^+ reduction and NO_3^- increase over a similar time period from a soil restored after stockpiling. In this study, however, losses through the drainage system were relatively small, compared to the total amount of N lost from the topsoil. This can be attributed to much lower rainfall (91 mm) than the long-term average (235 mm), (Smith and Trafford, 1976) which resulted in <1 mm of surface runoff and only 46 mm of drainage being recorded during this period.

From the end of October 1991 to the end of February 1992, total-N in the topsoil (0–0.25 cm) decreased rapidly (Fig. 2c) as a result of renewed mineralization and nitrification with subsequent leaching of mineral and possibly also soluble/particulate organic-N occurring at a relatively rapid rate. The increased rate of mineralization is likely to have been partly a consequence of a large increase in the size of the aerobic bacteria population (Williamson and Johnson, 1991). In total, 2100 kg ha^{-1} ($P < 0.01$) of mineral and organic-N was lost from the topsoil between October 1991 and February 1992. Harris and Birch (1989) estimated Kjeldahl N losses at 675 kg $\text{N ha}^{-1} \text{ yr}^{-1}$ for similar soils that had been stored. To

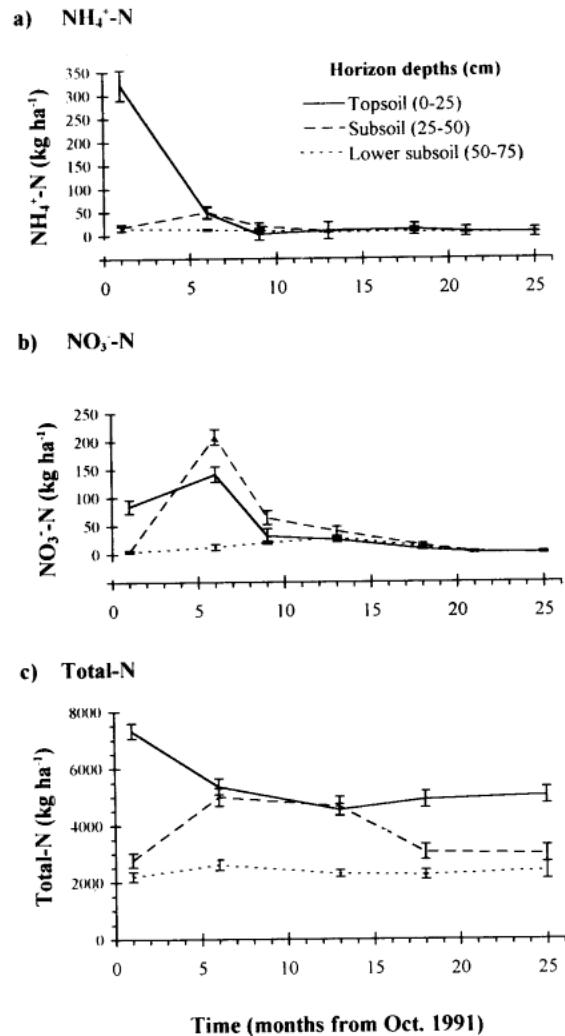


Fig. 2. Soil N at Butterwell from October 1991 to October 1993.

account for the very large loss of organic N in this study, N would have needed to have been mineralized and leached at a constant rate (assuming moisture and temperature permitted a constant rate) of approximately 13 kg $\text{N ha}^{-1} \text{ d}^{-1}$ between October 1991 and February 1992. This constant rate of mineralization and leaching was probably unlikely and therefore mineralization and leaching would have needed to occur at rates far in excess of 13 kg $\text{N ha}^{-1} \text{ d}^{-1}$ on days of favorable conditions. This may have been a possible route for the loss of some of the organic N but in combination with the movement of soluble/particulate organic N. A large labile pool of organic N was possibly created as a result of anaerobic conditions during soil storage (J. Harris, Environment and Industry Research Unit, Univ. of East London, 1995, personal communication) and the precise mechanisms

that led to the release of much of this N, and the transformations it underwent once released from its usually relatively inert state in an undisturbed soil (Addiscott et al., 1991) were not investigated.

Data collected for the subsoil at 0.25 to 0.5 m gave a strong indication that the 2000 kg of N that had been lost from the topsoil, and which had not reached the water-courses, had possibly entered the subsoil (Fig. 2c). The relatively small amount of the N in the subsoil occurring as mineral N at this time (Fig. 2a and b) may suggest that a large quantity of soluble/particulate organic N originating from the topsoil had been leached into the subsoil and/or immobilization of N, which was transported from the topsoil as NO_3^- , had then assimilated into the microbial population of the subsoil. Quantities of nutrients retained within the microbial community may be substantial and, under steady-state conditions, the population densities of soil microorganisms are relatively constant (Tate, 1987). However, after soil physical disturbance, and the possible release of carbonaceous nutrients from previously unavailable sites in one horizon, microbial biomass may be synthesized above the steady-state level in a lower horizon. A study that examined the rate of immobilization in an undisturbed soil (Ocio et al., 1991) suggested that the quantities immobilized during the period of this study (2000 kg ha⁻¹) would not be possible which supports the hypotheses that some of the N in the subsoil was leached from the topsoil and remained as soluble/particulate organic N in the subsoil.

To account for such a large amount of N leaching into the subsoil compared to the losses in drain water a two-phase system is invoked. Firstly, the small amount of water reaching the drains was, in all probability, moving partly as interflow through the topsoil and partly as by-pass flow through the few subsoiler cracks connected to the drainage system. However, although relatively large fissures created by the recent subsoiling under dry conditions had allowed possibly considerable water movement into the subsoil, the lack of a natural soil structure to connect the "manmade subsoiling fissures" with the drainage system may have resulted in water, bearing organic and mineral-N, being held up in "dead-end fissures". Water could then permeate slowly into the blocks of subsoil between the fissures. Secondly, capillary water movement into the subsoil will have occurred through the smaller pores. Wetting of the subsoil would be enhanced by the low gradients on the site that would have contributed to the movement of the majority of the N to the subsoil rather than it being lost via interflow to the drains. The quantity of water reaching the subsoil was still not sufficient for its return to field capacity.

Rainfall, drain discharge rate, and drain discharge NO_3^- -N concentration during the winter of 1991 and 1992 are shown in Fig. 3. Concentrations of NO_3^- -N observed in drainage water were well in excess of the levels defined in the European Community (EC) drinking water directive (11.3 mg L⁻¹). Initial samples of drainage water from some plots were <100 mg L⁻¹ but increased rapidly. Peak N concentration increased in each subsequent drainage event reaching a peak of 270 mg L⁻¹ in

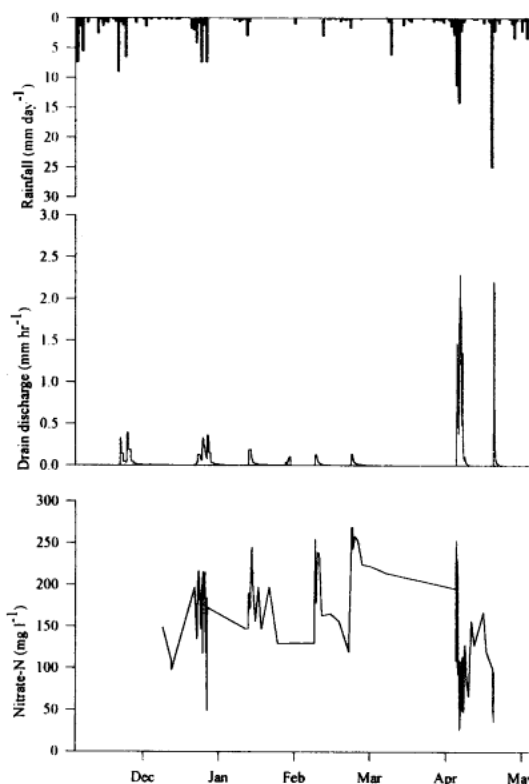


Fig. 3. Rainfall, drain discharge flow rate and drain discharge NO_3^- -N concentration, 1991/1992.

February. During March, insufficient flow took place to initiate any samples.

Spring and Summer 1992

From March to October 1992, 1000 kg ha⁻¹ of N was lost from the soil profile. Of this total, 59 kg ha⁻¹ was in drain water and surface runoff as mineral-N, up to 300 kg ha⁻¹ was estimated (not measured but typical for the soils and the region, Restoration of Land after Opencast Mining, 1984, Report 2, NCBOE Northern Research Project, 1984, unpublished data) to have been taken up by the grass crop and the remaining 641 kg ha⁻¹ apparently occurred as gaseous loss or leaching of soluble/particulate organic-N from both topsoil and subsoil (Fig. 2c).

Intensive rainfall during the period 30 Mar. to 7 Apr. 1992 generated drainflow equivalent to more than 50% of that recorded during the winter of 1991 and 1992, causing considerable losses of NO_3^- from the soil profile (Fig. 2). During this period, concentrations of NO_3^- -N in drain water remained above 50 mg L⁻¹ with peak concentrations of up to 250 mg L⁻¹ being observed. Towards the end of this period of drainage, the concentration observed started to drop despite the fact that losses were small in relation to the amount that remained in the subsoil (Fig. 2).

Only small amounts of surface flow were recorded and, as a consequence, samples were collected only from two of the four plots, for which the maximum concentration of NO_3^- -N was 171 mg L^{-1} . Because of the small volume of surface flow recorded this provided an insignificant route for losses of mineral-N compared to drainflow.

Concentrations of NH_4^+ -N in drain water typically remained around 0.2 mg L^{-1} rising to a maximum of 6.3 mg L^{-1} on occasions of peak drainflow. These concentrations are high in relation to the EC directive on drinking water that suggests a minimum acceptable concentration (MAC) of 0.5 mg L^{-1} with an average of 0.05 mg L^{-1} . This is much higher than has been observed in studies undertaken on undisturbed soils. At Brimstone Farm, a value of 0.5 mg L^{-1} was rarely exceeded with 90% of values being $<0.1 \text{ mg L}^{-1}$; when higher concentrations, in the range 0.9 to 1.2 mg L^{-1} , were observed these were associated with the movement of soil particles (Dowdell, 1984). In a study at Cockle Park on similar, but undisturbed, soil to that at Butterwell, (Hodgkinson et al., 1992), more than 90% of samples contained $<0.1 \text{ mg L}^{-1}$ under both arable and grassland cultivation.

Over these and subsequent periods, there was no increase in the total-N content of the lower subsoil (Fig. 2c); which supports the hypothesis that large gaseous losses and/or losses of soluble/particulate organic N had occurred. Moist soil conditions with spells of high rainfall, recorded during the period, may have been ideal for denitrification (Latey et al., 1981), even though the soil was not at field capacity. Soil temperature, although suboptimal, (Bouwman, 1989) may have been sufficient for a temperate soil (Powlson et al., 1988), when combined with the particular moisture conditions, to permit very high rates of denitrification. The pH was near optimum (Sahrawat and Keeney, 1986) and the readily decomposable pool of labile organic matter and microbial and abiotic biomass (Tate, 1987), will have been sufficient to facilitate extremely high rates of denitrification. Soil temperature rose to relatively high levels in late May while the soil was still near to field capacity, after which considerable drying occurred. During the period from 1 Mar. 1992 to approximately the end of May 1992 while the soil was wet, at an average temperature of 10°C and receiving intermittent and sometimes heavy rainfall, denitrification would need to have occurred at a daily average rate of nearly $7 \text{ kg ha}^{-1} \text{ d}^{-1}$ for the entire period to account for the measured total-N losses. This is exceptionally high when compared to a rate of $2.7 \text{ kg ha}^{-1} \text{ yr}^{-1}$ for a similar soil with similar management that had not been stored and reinstated (Eggington and Smith, 1986). However, because of the compaction that occurs during soil handling restored soils tend to have a greater number of small pores than undisturbed soils. These may have remained saturated for extended periods creating anaerobic zones between the larger fissures through which drainage occurred.

Between May and September 1992, relatively minor changes occurred in soil N at all depths as a result of soil conditions remaining dry.

Summary of Changes Occurring October 1991 to October 1992

To summarize N transformations and movements in the period October 1991 to October 1992, more than 2000 kg ha^{-1} of N was leached from the topsoil into the subsoil during the winter, with minimal crop uptake occurring. During the spring and summer, up to approximately 300 kg ha^{-1} was taken up from the profile by the crop, 59 kg ha^{-1} lost to the drains or runoff as mineral-N and the remaining 641 kg ha^{-1} is postulated to have either entered the atmosphere and/or entered the drainage system as soluble/particulate organic-N. On the study site, the 300 kg ha^{-1} of N estimated to have been taken up by the crop was only a temporary loss from the profile as the crop was flailed and ploughed back into the soil in the autumn. Little drainage occurred, partly because the soil did not reach field capacity and possibly because the nature of the fissures and pores in the restored soil did not allow adequate lateral flow to the drains.

Large amounts of N reached the subsoil compared to that lost in the drains. It is suggested that wetting up of the soil profile at depth occurred partly by water moving through the large fissures created by subsoiling that did not connect with the drainage system and partly through smaller pores carrying with it organic and/or mineral-N. Thus, a two-phase system of water movement may have been operating, which could have resulted in the depletion of NO_3^- -N in drainage water.

October 1992 to October 1993

During the hydrologic year 1992/93 the recorded rainfall of 641 mm was close to the long-term average of 670 mm (Smith and Trafford, 1976). However, the distribution of rainfall during the year did not follow the average pattern. Rainfall during the period up to the end of January was close to average, but March and February were much drier than normal with 9 and 13 mm being recorded compared to 49 and 41 mm , respectively. In contrast, April and May were much wetter than average with 96 and 116 mm being recorded respectively compared to long-term averages of 41 and 44 mm .

Autumn and Winter 1992/1993

From October 1992 to February 1993, 1500 kg ha^{-1} of N was lost from the subsoil horizon (0.25 – 0.5 m), this is shown in Fig. 2c. Only 30 kg of this was accounted for as mineral-N in the drainage and again only minimal crop uptake occurred with none moving to greater depth.

Compared to the previous year, the amount of surface flow that was recorded increased considerably, having become the major route by which excess rainfall left the site. The water balances for two of the plots indicated problems with ingress of foreign water during this period. Averaging the figures for the other two plots indicates that the combined discharge (surface runoff + drainflow) for 1992 and 1993 was 255 mm , of which just over 50% was surface runoff. This suggests that the fissures created by the subsoiling operation that had been carried out in 1991 had closed up, reducing the ability of the soil to

accept and transmit water. Visual examination of the topsoil during prolonged rainfall events suggested saturation readily occurring. An effect of this change in flow path on soil N could be to protect any N in the soil from leaching, since much of the rainfall left the site over the soil surface rather than through the soil profile.

It is suggested that the majority of drainage was occurring as interflow and bypass flow through the few larger fissures created by subsoiling that still remained, with little or no movement taking place below subsoiling depth (450 mm). In between these drainage paths there would be zones containing water and N that was less mobile, these zones would become anaerobic. For this model to be consistent with the large amount of gaseous loss that is hypothesized then these zones must only have been intermittently anaerobic; since substantial mineralization of immobilized N must have occurred to account for the amount of denitrification required to match the losses. An alternative model would be that the fissures created in the subsoil had closed up completely, and that water was moving through the topsoil, then along the subsoil/topsoil interface into the drains, the subsoil itself forming a separate intermittently anaerobic layer that did not interact with the drains.

At the commencement of drainage, the NO_3^- -N concentration in drain water was high (Fig. 4). This suggests

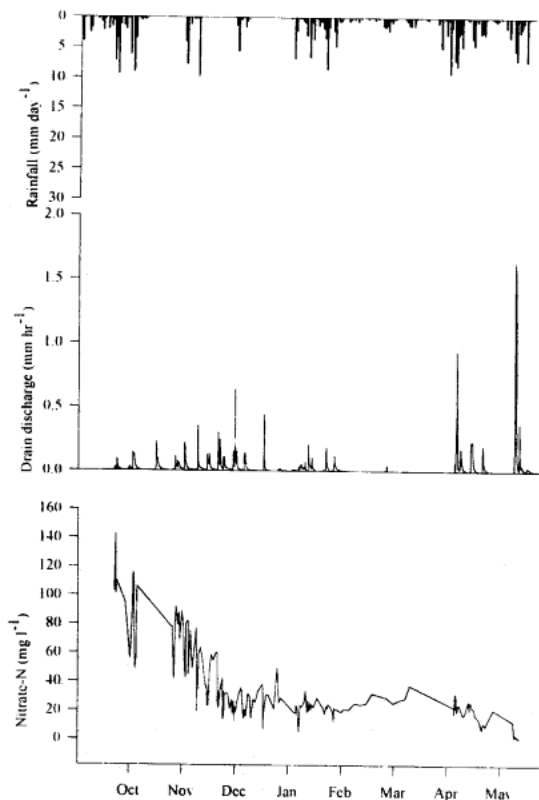


Fig. 4. Rainfall, drain discharge flow rate and drain discharge NO_3^- -N concentration, 1992/1993.

that fresh NO_3^- -N had become available in the soil adjacent to the routes through which drainage was taking place. The concentration of NO_3^- -N in the drainage water decreased steadily, as further drainage occurred, indicating that the amount available for leaching was being depleted. Under these circumstances, the fact that a large amount of water was moving as surface runoff will not have influenced the magnitude of NO_3^- -N losses in drainage water; since if more water had been passing through the soil profile, depletion of the zone adjacent to the routes by which drainage was taking place would have been more rapid. As soil measurements indicated that major N loss was occurring from the subsoil at this time and the fact that NO_3^- -N concentrations in drainage water steadily decreased, supports the proposition that some form of two-phase system may have been operating, leaching mineral and/or soluble/particulate organic-N.

During this period, concentrations of NH_4^+ -N in drainage water were typically $<0.1 \text{ mg L}^{-1}$, but, on occasions, short-lived concentrations as high as 5 mg L^{-1} were recorded, these can be attributed to transport of NH_4^+ in the particulate form.

The relatively wet and warm periods recorded in the autumn and winter of 1992/1993 would suggest that denitrification could have occurred at a very high rate. To account for the total loss of N, a constant rate of N loss over the entire period from October 1992 until February 1993 of $12.5 \text{ kg N ha}^{-1} \text{ d}^{-1}$ would have been necessary. However, although the soil was at moisture contents above field capacity, temperature in the subsoil in part of December, January, and February was below 5°C and therefore mineralization and denitrification were likely to have been relatively insignificant (Sahrawat and Keeney, 1986). This would mean that all N loss may have occurred on days when the soil temperature was above 5°C . Such days were between 25 October when the samples were taken and 15 December, after which the soil temperature finally fell below 5°C for the remainder of the winter sampling period. This amounted to 1472 kg ha^{-1} of N lost ($P < 0.01$) possibly in 45 d at denitrification rates more than $32 \text{ kg N ha}^{-1} \text{ d}^{-1}$. An associated increase in NH_4^+ -N and NO_3^- -N may have occurred within the period but were not detected in February by which time all mineral-N may have been lost from the soil. Such high daily rates of denitrification have been found to be potentially able to occur from similar stagnogley soils of the Denchworth series under ideal laboratory conditions at 20°C (Colbourn et al., 1984). In the field however, Colbourn et al. (1984) recorded denitrification rates of $<1\%$ of the potential measured in the laboratory. Consequently, for this study, a more plausible explanation might be that high denitrification rates may have occurred but in conjunction with losses of soluble/particulate organic N.

The soils reinstated for this experiment are common in northern England (Jarvis et al., 1984) and have been and are continuing to be widely disturbed for mining operations. Also the techniques used for restoration reported here are typical and commonly used. It would not therefore be unreasonable to expect similar rates of N loss to those found in this study to occur from most

similar soils that are stored for periods of more than 2 yr (Williamson and Johnson, 1991) and then reinstated. This would represent from a sample of 20 sites currently in operation for the next 12 yr in England, Wales and Scotland (British Coal Opencast, 1993), 1000 tonnes of N lost to the atmosphere and/or water courses per year.

Spring and Summer 1993

From 1 Mar. to 25 Oct. 1993, fluxes of N in the topsoil and subsoil had returned to levels similar to those in other undisturbed soils in the region (Fig. 2).

CONCLUSIONS AND RECOMMENDATIONS

The consequences of physical disturbance to the soil during stripping, stockpiling, and reinstatement were to cause unusually large N transformations and movements. In total, over the period from October 1991 to March 1993, 2600 kg ha⁻¹ of N was lost from the soil profile. More than 90% of this was unaccounted for in the soil and must therefore have been lost either to the atmosphere, possibly via denitrification as a combination of nitrous and nitrogen oxides and molecular N, or to the aquatic environment as mineral and soluble/particulate organic-N. The final fate of this N in the atmosphere or in the aquatic environment is unknown. This study served to indicate that apart from the depletion of a major soil nutrient there exists the potential for significant environmental pollution to either water or air over a period of some months following the reinstatement of stockpiled soil.

Further research is required to determine whether the N losses observed on this site are typical and what form they occur in. The proportion of gaseous losses occurring as either nitrous and nitrogen oxides or molecular N and the soil structural conditions associated with the method of loss will be of particular importance. Also, determination of microbial-N biomass directly by fumigation-extraction would further indicate the precise fate of N during this critical period and aid formulation of specific recommendations and further experimentation. Losses of soluble/particulate organic-N to the aquatic environment need to be quantified and subsequently its transformation after leaving the soil examined.

Such N losses might be reduced by preventing the development of anaerobic conditions in the soil mound. Soil storage for very short periods, periodically opening up and aerating the soil while stockpiled or permanently aerating and allowing drainage with a network of pipes are operations that may in part ameliorate the problem. However, this may be impractical in terms of cost and feasibility and could exacerbate structural damage incurred during soil movement. Post-restoration strategies to reduce N loss are an alternative, which possibly may be equally difficult to achieve. Soil management and drainage could be altered to encourage leaching, and the water draining from such sites would then require treatment to remove mineral and organic-N before discharge. Research is required to evaluate the efficacy of these proposed treatments.

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