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Nitrogen and sulphur management: challenges for organic sources in temperate agricultural systems

A.M. McNeill[1,*], J. Eriksen[2], L. Bergström[3], K.A. Smith[4], H. Marstorp[3],
H. Kirchmann[3] & I. Nilsson[3]

Abstract. A current global trend towards intensification or specialization of agricultural enterprises has been accompanied by increasing public awareness of associated environmental consequences. Air and water pollution from losses of nutrients, such as nitrogen (N) and sulphur (S), are a major concern. Governments have initiated extensive regulatory frameworks, including various land use policies, in an attempt to control or reduce the losses. This paper presents an overview of critical input and loss processes affecting N and S for temperate climates, and provides some background to the discussion in subsequent papers evaluating specific farming systems. Management effects on potential gaseous and leaching losses, the lack of synchrony between supply of nutrients and plant demand, and options for optimizing the efficiency of N and S use are reviewed. Integration of inorganic and organic fertilizer inputs and the equitable re-distribution of nutrients from manure are discussed. The paper concludes by highlighting a need for innovative research that is also targeted to practical approaches for reducing N and S losses, and improving the overall synchrony between supply and demand.

Keywords: Animal wastes, manures, excreta, plant residues, biological nitrogen fixation, volatilization, leaching, soil organic matter

INTRODUCTION

Agriculture has undergone drastic changes over the last four to five decades, and farms have increasingly become more specialized and more intensive, with livestock and arable production often being concentrated in different regions of a country. These changes have been accompanied by a greater potential for losses of nutrients, such as nitrogen (N) and sulphur (S), to the environment. Dissolved N and S chemical species (mainly NO_3^- and SO_4^{2-}) may be leached to surface and/or groundwaters, or lost through gaseous emissions of ammonia, nitrous oxide (from microbial nitrification and denitrification) and volatile organic S compounds such as sulphides. Increased awareness of these environmental effects, coupled with the changes in farm systems, has stimulated extensive governmental frameworks that are attempting to control or reduce the losses through various regulations and land use policies.

Currently, there is another trend – primarily in North American and European farming systems, but also in China, Latin America and Africa – towards on-farm recycling of animal manures or increased use of organic fertilizers that could lead to more sustainable nutrient management (Stockdale *et al.* 2001). This shift towards organic farming practices has received political attention in Europe because the general public perceives that these practices reduce environmental problems related to intensive agriculture (National Research Council 1989; Granstedt 1995). As an example, the Swedish parliament has passed a law stating that by the year 2005, 20% of the agricultural land should be under organic farming. However, use of organic fertilizers on a large scale is not without risk. Even with best management practices there are new challenges associated with organic fertilizers, because the risk of losses to water and air is potentially larger, and the use efficiency is lower, compared with the use of equivalent quantities of inorganic fertilizers (Bergström & Kirchmann 1999).

Nitrogen and sulphur are essential plant nutrients that are often a focus for this nutrient management debate because (i) the anionic forms (NO_3^- and SO_4^{-2}) of both are mobile and easily lost from many soils through leaching, and (ii) the overwhelming proportion of both nutrients in the soil is found in an organic form and not immediately available to plants. Therefore, management

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[1]Soil & Land Systems, School of Earth & Environmental Sciences, University of Adelaide, Roseworthy Campus, SA5371, Australia. [2]Department of Agroecology, Danish Institute of Agricultural Sciences, PO Box 50, 8830 Tjele, Denmark. [3]Department of Soil Sciences, Swedish University of Agricultural Sciences, Box 7014, SE-750 07 Uppsala, Sweden. [4]Institute of Atmospheric & Environmental Science, School of GeoSciences, University of Edinburgh, Darwin Bldg, Mayfield Rd, Edinburgh EH9 3JU, UK.
*Corresponding author. Fax: +61 8 8303 7979.
E-mail: ann.mcneill[a]adelaide.edu.au

solutions affecting one nutrient will also, potentially, have an effect on the other. Also, over the last few decades, industrial Western societies have reduced atmospheric pollution by replacing high-S coal and oil with low-S natural gas, nuclear energy, and renewable energy sources. These changes in fuels, combined with the use of fertilizers with lower by-product S, have resulted in a major reduction in S deposition and application to the land. The reduced inputs, coupled with cropping system changes that have resulted in growing crops with a greater S requirement, for example oilseed rape (*Brassica napus* L.), have thus created a first-time need for S fertilization in some soils.

The multitude of interactions among agricultural practices (e.g. crop sequence), industrial practices (e.g. fuel sources), and policy decisions (e.g. organic versus inorganic farming systems) means that for increased sustainability it is essential to consider how all factors may be affected before adopting any one solution. In that context, this paper presents quantitative information concerning the key input and loss processes for N and S in temperate agricultural systems. The focus is primarily on organic nutrient sources because currently less information is available on those practices than on inorganic ones (Watson *et al.* 2002). Potential management solutions that can efficiently utilize both inorganic and organic N and S sources for crop and pasture production while minimizing losses to the environment are reviewed, although greater detail will be provided in the subsequent papers of this Supplement dealing with overall nutrient management for specific farming systems.

INPUTS AND RECYCLING OF NITROGEN AND SULPHUR

Atmospheric deposition

Deposition of N can vary between 5 kg N $ha^{-1} yr^{-1}$ in northern Scandinavia and 45 kg $ha^{-1} yr^{-1}$ in the Benelux countries (Ferm 1998). In Western Europe and other intensively farmed regions, most N that is deposited has previously been emitted as ammonia (NH_3) from manure (faeces and urine) of agricultural livestock (ECETOC 1994), or as NO_x compounds from vehicular traffic and stationary combustion (Dentener & Crutzen 1994). Emission and deposition of ammoniacal N in Europe is still considerable, despite measures to reduce NH_3 volatilization from agriculture. Such measures include covering slurry and urine storage facilities, rapid incorporation of animal wastes after spreading on bare soil, development of new spreading techniques, and improved handling and ventilation within animal houses (Gustavsson 1998). Furthermore, because all atmospheric N compounds are water-soluble and easily taken up by organisms, deposition readily leads to eutrophication, acidification and/or damage to sensitive types of vegetation, for example natural ecosystems such as woodland, heathland or grassland, if the critical load is exceeded.

In contrast to high N deposition, sulphur deficiencies have been reported in previously S-sufficient areas. The main reasons for this include: (i) the environmental control of SO_2 emissions in industrial areas and ensuing declines in S deposition, (ii) the increasing use of P fertilizers with a low S content, (iii) the increase in yields obtained as a result of other technological improvements, and (iv) the decreasing use of S-containing pesticides (Blair 2002). In Europe, concentrations of SO_2 in the atmosphere have decreased dramatically during the last 20–30 years (Figure 1), leading to decreased deposition of S on agricultural land. Current total S deposition calculated by EMEP (Programme for Monitoring and Evaluating of Long-range Transmission of Air Pollutants in Europe) ranges from 1 kg S $ha^{-1} yr^{-1}$ in rural areas of northern Norway to more than 20 kg $ha^{-1} yr^{-1}$ in industrial areas, with an average of 2–10 kg S ha^{-1} in many areas (McGrath *et al.* 2002).

Legume nitrogen fixation

Inputs of biologically fixed N to temperate farming systems occur largely via legume–rhizobium symbioses, with relatively insignificant additions from non-symbiotic and

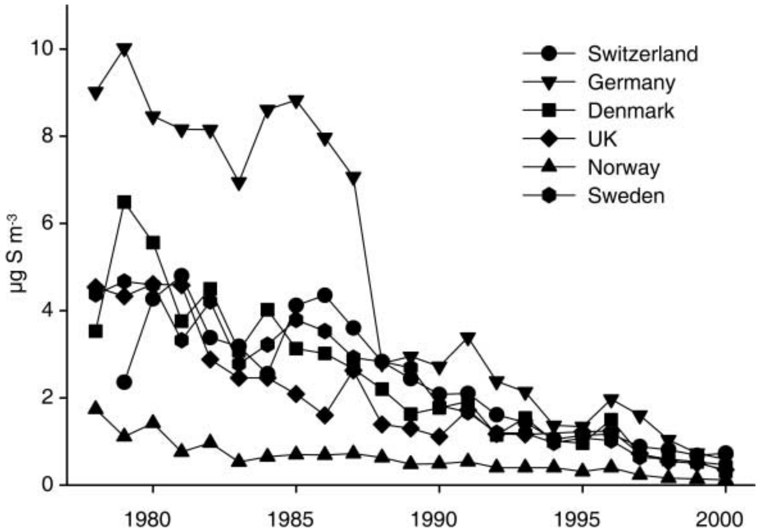

Figure 1. Atmospheric sulphur dioxide concentration at different sites in Europe. (Source: NILU 2003.)

associative sources (Chalk 1991; Kennedy & Islam 2001). The potential for N fixation within temperate farming systems ranges from 200 to 700 kg N ha^{-1} yr^{-1}, depending upon the type of legume (Ledgard 2001), although the potential is often not achieved and values as low as 2 kg N ha^{-1} yr^{-1} have been reported (Peoples *et al.* 1995).

Overall, inputs of fixed N are governed by the productivity of the legume as influenced by management, climatic and edaphic conditions and the dependence of the legume on N fixation (Van Kessel & Hartley 2000). The latter is primarily determined by the effectiveness of the host legume–rhizobium symbiosis and has been reported to vary widely both within and between legume species (Peoples *et al.* 1995; Jensen 1997; Unkovich & Pate 2000; Peoples & Baldock 2001). Published field data are relatively scarce, particularly for cool temperate regions, but Australian studies suggest that fixation is often lower than the optimum due to absence of suitable rhizobium strains or inhibition of an effective symbiosis by elevated N fertility status of the soil (Schwenke *et al.* 1998).

Nitrogen inputs from grain legumes are highly dependent on the crop N harvest index, that is the proportion of total above-ground N production removed as grain. Data from temperate Europe and Asia, North America and Australia (Beck *et al.* 1991; Peoples *et al.* 1995; Jensen 1997; Van Kessel & Hartley 2000) show an enormous range in this variable, and it is not surprising that some studies show a positive effect of grain legumes on the soil N balance (Badaruddin & Meyer 1994; White *et al.* 1994; Schwenke *et al.* 1998), whereas others show a neutral or a negative effect (Beck *et al.* 1991; Hossain *et al.* 1996; Armstrong *et al.* 1997). However, most of these N balances have probably underestimated the below-ground input of fixed N by legumes due to problems of root sampling and quantifying root exudates or rhizodeposition (Russell & Fillery 1996; Jørgensen & Ledgard 1997; McNeill *et al.* 1997; Unkovich *et al.* 1997; Rochester *et al.* 1998; Papastylianou 1999; Khan *et al.* 2003; Mayer *et al.* 2003).

Quantifying inputs of fixed N by pasture and forage legumes is difficult, owing to the recycling that occurs through the grazing animal and the N transfer between legume and non-legume species in the system. The fixation is also regulated by a feedback mechanism involving the soil inorganic N content and competition from associated grasses (Ledgard & Steele 1992). Field studies report fixation rates from as low as 2 kg N ha^{-1} yr^{-1} up to 315 kg N ha^{-1} yr^{-1} (Vinther & Jensen 2000; Ledgard 2001; Peoples & Baldock 2001). Clearly, including estimates of fixed N partitioned below ground (as mentioned previously) would increase the magnitude of these estimates.

Nitrogen fixed by legumes represents a key contribution to nutrient cycling in legume-based farming systems by increasing soil organic N status, although the process may also contribute to an elevated nitrate leaching potential on certain soil types if not managed effectively.

Plant residues and soil organic matter (SOM)

The magnitude and timing of N and S release from residues, and any subsequent immobilization, are strongly influenced by the efficiency of carbon (C) use by the decomposer population, the demand for N and S, the chemical nature of the plant residues and a range of soil factors (Kumar & Goh 2000). For non-leguminous crops, return of residues '*in situ*' is not strictly an input but can be considered 'recycling' of nutrients, extracted from, and then returned to, the labile SOM pool (Jenkinson 1990; Parton *et al.* 1994). Subsequently, the organic N and S may enter either a more stable organic matter pool or an inorganic (plant-available) pool.

Immature or 'fresh' plant materials, such as those in green manures, primarily contain sulphate (up to 10% of total S content) and nitrate (up to 5% of total N content) that may be very rapidly released and available for plant uptake (Whitehead 2000). These 'fresh' or green residues tend to decompose rapidly, with up to 40% of residue mineralized after one year. Mature residues decompose more slowly than 'fresh' residues because of a wider C/N ratio and greater lignin/N or polyphenol/N ratios. The impact of these differences in residue decomposition rates on N and S cycling is that 10–20% of N in green residues will typically be utilized by the immediate or succeeding crop, whereas $<10\%$ of the N in mature residues is taken up (Fillery 2001 and references therein). The primary sink for most of the N in legume residues is the SOM pool; nevertheless, research has clearly highlighted that N released from legume residues needs to be synchronized with the demand for N by the following crop (Myers *et al.* 1994, 1997) in order to prevent substantial leaching losses of legume-derived nitrate under wet temperate conditions (see later section on loss processes) or after fallow periods.

Animal manures

On a dry matter basis, N and S concentrations in animal wastes amount to 1.9–10% N (mean: 3.2% N) and 0.6–0.7% S (Kirchmann & Witter 1992; Steineck *et al.* 1999). This is equivalent to 70–80% of the amount present in fodder. The N and S contents of stored manures can vary, as often they are mixtures of excreta of different age from animals fed different diets, and possibly come from different animal species. Slurries may contain 2.8–5.1 kg N m^{-3} and 0.15–0.7 kg S m^{-3} (Eriksen *et al.* 1995), and manures 4.8 – 7.7 kg N $t^{-1}$and 0.69–1.4 kg S t^{-1} wet weight (Steineck *et al.* 1999). On a global scale, S excretion from farm animals is estimated to be around 8 million metric tonnes per year, corresponding to 80% of the world consumption of mineral S fertilizer (Eriksen 2002).

The conversion of dietary N into animal protein typically ranges from 5 to 20% and is generally lower for legumes than grasses (Ulyatt *et al.* 1988). Balancing the supply of N in animal diets can be easily achieved by using legumes or well-fertilized non-legumes as fodder, but the intake of S by animals may often be limited by a low S content of the fodder. For example, in diets for monogastrics where cereal grains constitute the main source of protein, methionine is often one of the most limiting essential amino acids (Heger *et al.* 1997). Also, cattle diets in areas with low S deposition may be short of S (Aaes & Thøgersen 1999). Under such conditions, the S content of the manure will be relatively low and mainly in the organic form, which is not readily available to crops after soil application. Furthermore, more efficient fodder utilization may also affect amounts of N and S excreted per unit of animal or milk produced, which may affect the composition of animal wastes. A greater portion of N and S will probably be present in organic forms in animal wastes and less will be present in urine (Kyvsgaard *et al.* 2000).

Annual application of manure will increase the soil organic N and S content in the long term, although the extent of this increase depends on soil type, cropping system and management. Manure applications may also increase the potential mineralization rate. Therefore, a residual effect of long-term organic manure application on the capacity of the system to supply plant-available N should be expected, although there is no evidence that either organic N or organic S in manure will mineralize more readily than the bulk of soil organic N or S. The ability of a cropping system to utilize mineralized N and S will depend on the length of the growing season, although (as mentioned in the previous section) mineralization is unlikely to fully match crop demand.

LOSS PROCESSES FOR NITROGEN AND SULPHUR

Volatilization

Manure may be deposited by grazing livestock directly on to pastures, or spread on the land surface as solid farmyard manure (FYM) or liquid slurry, and often can correspond to an annual application rate of several hundred kilograms of N per hectare. Following excreta deposition or manure spreading, the fraction of N lost depends on several environmental and edaphic factors. The rates of hydrolysis of urea and dissociation of NH_4^+ to dissolved NH_3 and H^+ are both temperature-dependent, as is the rate of diffusion of dissolved NH_3 to the interface with the atmosphere. This means that the rate of emission of NH_3 to the atmosphere increases substantially as the ambient temperature increases. Other factors include: the micrometeorological conditions affecting the rate of transfer of ammonia from the land surface into the atmosphere; the depth of application; within the soil; the initial pH (losses are inevitably greater in soils that are already alkaline); the buffer capacity of the soil; and the soil water content (e.g. Yan *et al.* 2003).

Ammonia emissions from land-spreading of solid, straw-based FYM from cattle and pig housings can be of the order of two-thirds of the total ammoniacal nitrogen (TAN) content (Chambers *et al.* 1997; Menzi *et al.* 1997; Misselbrook *et al.* 2000). Similar losses from liquid slurries spread on the soil surface have been reported (Huijsmans *et al.* 1997; Misselbrook *et al.* 2000). Ammonia losses increase with increasing dry matter (DM) content of the slurry (Jarvis & Pain 1990; Sommer & Olesen 1991), within a range of 1–9% DM for applications in the autumn–spring period (Smith & Chambers 1995). The emissions from ammoniacal and uric acid N in poultry manures are substantially lower than those from cattle and pig manures (Chambers *et al.* 1997; Menzi *et al.* 1997).

During storage of animal wastes in slurry tanks, urine pits and solid manure heaps or piles, anaerobic microbial processes dominate, whereas aerobic decomposition prevails only in well-aerated composts and deep litter. Formation of volatile ammonia or ammonium from organically bound N is a main change during anaerobic storage (Kirchmann & Witter 1989). Similarly, S present as SO_4^{2-} can be reduced and gaseous S compounds formed under anaerobic storage conditions. These compounds, such as carbon disulphide (CS_2), carbonyl sulphide (COS), dimethyl sulphide (CH_3SCH_3), dimethyl disulphide (CH_3SSCH_3) and methyl mercaptan (CH_3SH), as well as hydrogen sulphide (H_2S), can be lost through volatilization (Banwart & Bremner 1975). However, Beard & Guenzi (1983) reported minimal loss of S by volatilization from stored slurries where redox conditions were controlled. Another study by Eriksen *et al.* (1995) reported a decrease in the total SO_4^{2-} concentration (organic ester sulphates and SO_4^{2-}) and an increase in sulphide and C-bonded S in both cattle and pig slurry during storage from February to November. Therefore, as a consequence of storage, different slurries may be expected to have different levels of plant-available S and N.

The anaerobic conditions that are sometimes induced during application of large amounts of manures to fields can also result in production of volatile sulphides (mainly H_2S). However, these sulphides generally react with soil constituents such as Fe oxides to form FeS_2 or FeS (Metson 1979), which often results in no volatilization of H_2S from soils (Freney 1986), except when waterlogging is complete and prolonged, and trace amounts of carbon disulphide or dimethyl sulphide produced by microorganisms may volatilize (Farwell *et al.* 1979; Brown 1982). Janzen & Ellert (1998) found emissions of S from soil to have a mean rate of less than $0.2\,\mathrm{kg\,S\,ha^{-1}\,yr^{-1}}$ and from plants $0.1–3\,\mathrm{kg\,S\,ha^{-1}\,yr^{-1}}$.

Denitrification

Microbial denitrification – the anaerobic reduction of nitrate present in the soil to gaseous nitrous oxide (N_2O), nitric oxide (NO) and dinitrogen (N_2) – is the main process by which chemically or biologically fixed N is returned to the atmosphere. Estimates of the quantities of N lost by the denitrification mechanism from agricultural soils have varied widely. Hauck (1986) suggested losses of 20–40% of the N applied, whereas Nieder *et al.* (1989) gave figures for cropland ranging from 2.5% to more than 50%.

Nitrate is the principal substrate for denitrification, and thus is a key driving variable (Weier *et al.* 1993), since low concentrations of NO_3^- constrain denitrification rates. The soil water content (more particularly the water-filled pore space [WFPS] with its effect on soil aeration) is another major controlling factor. Diffusion of oxygen throughout the soil matrix is usually sufficient to prevent substantial rates of denitrification in soils with plenty of macropores, for example, in coarse sandy soils and in finer textured ones with good structure. However, where there is poor structure – with larger peds rather than small crumbs being common, or loss of air-filled macropores as a result of compaction by wheels, tillage machinery, or livestock hooves – the outcome is likely to be the development of anaerobic zones within some of the soil structural units, and an enhancement of denitrification (e.g. Nieder *et al.* 1989; Arah *et al.* 1991).

Soils with greater amounts of labile organic matter will respire faster than those with less, and thus the former have a greater potential for denitrification than the latter (Bijay-Singh *et al.* 1988). Webster & Goulding (1989) showed that plots with high C content, as a result of receiving large inputs of FYM over many years, showed much higher denitrification rates in the autumn than adjacent plots that had received only mineral fertilizers.

Recorded rates of denitrification are relatively few, because of the experimental difficulty associated with making quantitative measurements, but even so, values up to some hundreds (and, occasionally thousands) of grams of N per hectare per day have been reported (Arah *et al.* 1991; Weier *et al.* 1993). Losses exacerbated by the addition of plant residues have been identified in some studies. For example, on a clay loam soil in Scotland where plots were ploughed and sown to grass in July with an addition of 64 kg N ha^{-1} as $Ca(NO_3)_2$, about 18 kg N ha^{-1} were denitrified from the plots without residues, and twice that amount from those with residues (Vinten *et al.* 1996). There is other evidence that N-rich crop residues may stimulate rapid denitrification and associated nitrous oxide emissions, even in coarse-textured soil. For example, Velthof *et al.* (2002) reported that N_2O emissions range up to 14.6% of the N added as brassica and sugar beet residues, whereas emissions after adding cereal residues were not significantly different from the control. Similarly, Baggs *et al.* (2000) found that more than 1 kg N_2O-N ha^{-1} was released from a freely drained loamy sand over the first 14 days after incorporation of lettuce (*Lactuca sativa* L.) residues.

Leaching

Leaching of inorganic N as nitrate (NO_3^-), either applied as a fertilizer or derived from mineralization of organic matter inputs, is affected by climate, soil type, geo-hydrological conditions, crop species, timing and rate of fertilizer inputs, and soil management. Generally, NO_3^- leaching increases as the input of NO_3^--N increases, although the increase in leaching is usually not continuous but follows a linear split-line relationship with a distinct break-point (e.g. Bergström & Brink 1986; Lord & Mitchell 1998). From zero NO_3^--N addition up to optimum application rates, the increase in NO_3^- leaching is minimal, whereas at higher rates leaching often increases considerably.

Lord & Mitchell (1998) showed that as long as crop removal of N was 52% of that applied to cereal crops, which is the normal efficiency at economic optimum conditions, leaching was hardly affected at all by the NO_3^--N input. However, because of the uncertainty involved in predicting the break-point, it was concluded that a small reduction in N input below the expected economic optimum rate is desirable to considerably reduce the risk of high N-leaching losses. Another strategy is to use multiple, small fertilizer-N applications, although the long-term effects of differentiated N applications are still relatively unknown.

Application of large amounts of manure on agricultural fields certainly increases the risk of high N-leaching loads, particularly where drainage is an important component of the water balance. Even relatively modest application rates of manure N (e.g. 110 kg NH_4^+-N ha^{-1}) can result in N-leaching loads that are higher than if corresponding amounts of inorganic N fertilizers are used (Kemppainen 1995; Thomsen *et al.* 1997), particularly where mineralization and plant uptake are not synchronized. For example, in a study where an equal amount of N as manure or inorganic fertilizer (100 kg ha^{-1}) was applied to barley, leaching of N originating from manure was almost ten times higher than that of fertilizer N (Bergström & Kirchmann 1999). However, given best practice management in both cases, leaching loss under an organic system was reported as similar to, or slightly smaller than, that from a conventional system (Stopes *et al.* 2002).

Total N leaching losses in grazed systems vary, ranging from relatively small amounts (6–34 kg N ha^{-1}) on sheep-grazed temperate grass/clover pastures (Cuttle *et al.* 1998) to larger amounts (20–74 kg N ha^{-1}) reported for intensively grazed dairy cattle pastures (Ledgard *et al.* 1999). In one study, about 55% of the leached N was derived from the urine patches (Ruz-Jerez *et al.* 1995), probably because deposition of N in urine patches by grazing animals generally exceeds the capacity of the plants to utilize the inorganic N produced, particularly NO_3^- (Ball & Ryden 1984; Haynes & Williams 1993). Reports of the contribution of legume-fixed N to leaching losses also range widely from 8 to 80% (Anderson *et al.* 1998a, b; Ledgard *et al.* 1999; Ridley *et al.* 2001).

Evidence suggests that excess S inputs from atmospheric deposition in the past were leached as SO_4^{2-} (Eriksen 1996; Knights *et al.* 2000), particularly on certain soil types. The retention of SO_4^{2-} in soils depends on the nature of the charged mineral surfaces, the pH, and concentrations of SO_4^{2-} and other ions in the soil solution (Harward & Reisenauer 1966); furthermore, above pH 6 virtually all soil SO_4^{2-} is found in solution (Curtin & Syers 1990). Even in soils with mineralogy conducive to a high SO_4^{2-} retention capacity, the retention is weak at high pH, and SO_4^{2-} can be removed by repeated extractions with water (Chao *et al.* 1962). As a consequence, many agricultural soils are prone to SO_4^{2-} leaching, and the amount of leached SO_4^{2-} is closely related to the drainage volume (Shepherd & Bennett 1998; Eriksen & Askegaard 2000).

In a dairy crop rotation on a sandy soil, the average SO_4^{2-} leaching from the crop rotation was 20 kg S ha^{-1} yr^{-1}, equivalent to 60% of the total input to the rotation (Eriksen & Askegaard 2000). Sulphate leaching was very variable, ranging from 4 to 45 kg S ha^{-1} yr^{-1} for the same crop in different years. In a long-term field experiment on a clay loam in south-central Sweden, the average leaching during 35 years in plots treated with $Ca(NO_3)_2$, green manure and animal manure was estimated to be 24, 34 and 38 kg S ha^{-1} yr^{-1}, respectively, which corresponded to 65, 71 and 69% of the S inputs (Kirchmann *et al.* 1996). Obviously, reliable estimates of SO_4^{2-} leaching will be required when mass balances are used for determining the S status at field or farm level. This is especially important for temperate regions with a high winter rainfall and soils with a low SO_4^{2-} retention capacity.

OPTIMIZING USE EFFICIENCY OF NITROGEN AND SULPHUR

This section focuses on potential management options to improve the management of N and S in agricultural systems by optimizing inputs from legume N fixation, plant residues and animal wastes, while minimizing losses via volatilization, denitrification and leaching. Central to these management strategies is recognition of the lack of synchrony between nutrient release from organic sources and crop demand in current farming systems, and thus the potential requirement for integration of organic and inorganic fertilizer inputs. The concept of equitable re-distribution of nutrients from manure is also discussed.

Inputs from legume nitrogen fixation and plant residues

Management practices that can optimize inputs from legume N fixation include maximizing plant growth by providing adequate nutrition, especially P, ensuring the presence of an effective rhizobium strain, and minimizing pests and diseases. Also, in a rotation, sowing the legumes into soils with low inorganic N status can increase N fixation. Manipulating the mixture of non-legume and legume species within a pasture, for example by using appropriate grazing strategies, can also enhance legume N fixation.

Transfer of fixed N within a single season from legume to non-legume via rhizodeposition is recognized around the world as a vitally important component of nutrient cycling in low-input temperate pasture systems (Boller & Nösberger 1987; Ledgard & Steele 1992). Controlled environment studies have demonstrated that as much as 15% of total plant N in ryegrass can originate from N exuded from companion white clover (Paynel *et al.* 2001). Long-term studies in the field have produced results of a similar order of magnitude for older pasture plants (Soussana & Hartwig 1996; Høgh-Jensen & Schjoerring 2001), as well as for some grain legumes (Sawatsky & Soper 1991; Jensen 1996c), with substantial uptake by companion non-legume crop species (Jensen 1996a). The significance of transfer of fixed N under field conditions within a season, for temperate inter-cropping systems, has yet to be clearly defined; reports from the limited data available are mixed (Danso *et al.* 1987; Papastylianou 1988; Cowell *et al.* 1989; Izaurralde *et al.* 1992; Waterer *et al.* 1994; Jensen 1996b). Overall, the amount of transfer will, of course, depend on plant species and plant age, as well as on the abiotic and biotic environment (Fujita *et al.* 1992). It will also be influenced by defoliation (grazing or cutting), although quantifying the effects of such management practices on N transfer has proved difficult and further research is required (Ledgard 2001).

The asynchrony in supply and demand for N and S in organic systems is a major concern and may not be recognized by policy-makers advocating increased adoption of those practices. Current research has focused on manipulating residue quality and quantity, including additions of on-farm and off-farm waste products, to manage nutrient supply and availability to crops. The time course of net N mineralization has been affected by manipulation of the residue C/N ratio, or the phenol content, but also by varying types and concentrations of carbohydrates (e.g. Handayanto *et al.* 1997; Gunnarsson & Marstorp 2002; Rahn *et al.* 2003; De Neve *et al.* 2004). These practices are particularly critical for areas where leaching is likely (see section below on leaching). However, most approaches, while reducing the high N release, also reduced the total amount of available N in the long term (Handayanto *et al.* 1997). There is clearly a need to identify practices that both improve synchrony and can be adopted by farmers (Myers *et al.* 1997). In drier climates, this may include management strategies that ensure rapid root development and thus crop N uptake in response to the onset of rains. Other suggestions for exploiting resource capture include relay cropping with deep-rooted cover crop species (Giller *et al.* 1997), intercropping (Watson *et al.* 2002) and selection of crop varieties or types that are efficient in utilizing available nutrients (Foulkes *et al.* 1998, Le Gouis *et al.* 2000).

Managing nitrogen in animal wastes

As mentioned earlier, volatilization of NH_3 from animal wastes can occur during housing, storage, application to soil, or grazing (Bussink & Oenema 1998). Measures to minimize losses of NH_3 should lead to an improved utilization of manure N, but all parts of the manure handling chain need to be considered, since intervening in one part affects losses in another. For example, losses are more likely during aerobic storage of manures (composting) than after subsequent soil application, because the remaining N is mainly in organically bound N forms (>90%) and thus concentrations of inorganic N are very low (Kirchmann 1985). On the other hand, following anaerobic storage a significant part of the manure N is present as NH_4^+ and there is a potential for large gaseous losses when the manure is applied to soil. Kirchmann & Lundwall (1998) concluded that anaerobic waste management, which is standard agricultural practice in Europe, resulted in smaller ammonia losses overall than aerobic treatment. They reported a minimum of 10–15% of the total N present lost from the former, but at least 20% and up to 70% from the latter.

A major reduction in gaseous emissions from anaerobic wastes can be achieved by replacing simple surface-spreading techniques by other procedures, such as narrow-band application to the soil surface or shallow injection into open or closed slots (Huijsmans *et al.* 1997; Lorenz & Steffens 1997). Huijsmans *et al.* (2001) found that, on average, narrow-band application reduced emissions by 74% compared with those following surface spreading, and injection reduced them by 92% (Table 1). However, Pain & Misselbrook (1997) reported much smaller reductions in the UK.

Table 1. Cumulative NH_3 losses over 96 h after application of slurry to grassland by different methods. (Based on data of Huijsmans *et al.* 2001.)

Application method	NH_3 losses (% of NH_4-N applied)	
	Range	Mean
Surface spreading	27–98	77
Narrow band spreading	8–50	20
Shallow injection	1–25	6

Apart from these issues of N conservation during storage, handling and application of manure, the ultimate availability of the N for plant uptake is also important. This will clearly be a function of chemical composition. Inorganic N in anaerobic manure has been shown to be equally as plant-available as N from inorganic fertilizer, if the materials are applied at the same time of the year (Kirchmann 1989). Fatty acids present in anaerobic manures can cause a short-term, initial immobilization of N, but this does not seem to affect N utilization over a growing season (Kirchmann & Lundvall 1993). In contrast, organic N in aerobically stabilized manures is only available to crops after net N mineralization, which is not necessarily synchronized with plant demand (Myers *et al.* 1997). Thus, inorganic N derived from organic sources may potentially be lost via leaching or denitrification.

A more fundamental approach towards reducing ammonia emissions from livestock systems is to decrease N inputs, as opposed to dealing with the N-rich manure after it is produced. Hilhorst *et al.* (2001) showed that very high N utilization efficiencies in animal nutrition and crop production permitted a similar milk production to that of conventional intensive dairy farms, but at a much lower input level, while achieving a reduction in the annual NH_3 emission from 64 kg NH_3-N ha^{-1} on the average farm to only 20 kg NH_3-N ha^{-1}. Ledgard (2001) discussed the potential for improving N utilization by grazing animals and highlighted the need for more innovative approaches, such as dietary manipulation involving certain fodder species or supplements that alter the ratio of dung to urine in the excreta and thus reduce the potential for losses.

Management options to reduce nitrogen leaching

Preventing N leaching is most critical in areas with light-textured soils, in cold and humid climates and where geohydrological conditions allow for deep percolation of infiltrating water. In these regions, it is obvious that manures applied to bare soil in autumn pose a higher risk for leaching than when they are applied in spring. In particular, early applications immediately following harvest can increase N leaching several-fold compared with early winter/spring applications (Beckwith *et al.* 1998). The application rate of manure on individual fields, which will determine the risk for leaching, depends largely on the number of animal units on each farm. In Sweden, there are regulations limiting the stocking rate to 1.6 large livestock units per hectare, based on annual P removal by crops. However, preliminary estimates based on the above-mentioned study by Bergström & Kirchmann (1999) indicated that this number had to be lowered to 0.2 in order to reduce N leaching to the level obtained when 100 kg N ha^{-1} of inorganic N fertilizer was used. In Denmark, the number of livestock units is not regulated, but the total application of manure N and fertilizer N on each field is controlled instead.

In cold, humid regions, the problems associated with manures are caused by the fact that inorganic N is often released too late in the season, or after the growing season, and is therefore available for leaching. One solution to reduce this asynchrony between demand and supply of N is to modify the composition of the organic materials that are applied on fields. This can be achieved by adding organic materials with a high C/N ratio (e.g. paper-mill waste; Kirchmann & Bergström 2003) to the animal manures when they are incorporated into soil. Such practices lead to increased N immobilization during periods when the risk for N losses is high.

In recent years, cover crops or ‘catch crops’, have proven to be effective in reducing N leaching when grown in conjunction with crops that require high levels of N, such as maize (*Zea mays* L.) and other cereal grains (see reviews by Meisinger *et al.* 1991 and Aronsson 2000). The cover crop can take up substantial amounts of N after harvest of the main crop in autumn, and several studies around the world have demonstrated large consequent reductions in N leaching (Martinez & Guiraud 1990; Nygaard Sørensen 1991; Shepherd 1999; Bergström & Jokela 2001), in some cases by greater than 80% (Lewan 1994). However, there are still uncertainties on how decomposition and mineralization of cover crop residues that are incorporated into soil will affect leaching of N over the long term. Some will be mineralized very soon after incorporation and some will form quite stable fractions and enter the soil humus pool. Especially when growing cover crops repeatedly, this increase in humus may contribute to larger N leaching losses over the long term, even though the short-term effect is favourable. Nevertheless, there is reason to believe that cover crops will be used in the future to reduce N leaching effectively.

The tremendous spatial variability in soil properties and environmental conditions in a region needs to be taken into account when considering countermeasures for reducing N leaching (e.g. cover crops, minimum tillage). Thus, practices can be targeted to the most sensitive areas, and these may range from catchments to sub-catchments, whole farms to individual fields, and even parts of fields.

Management options for sulphur

Although the organic S fraction of soil and added manures may be important for the supply of S to plants in deficiency situations, it is often found that mineralization of soil organic S cannot supply the required amount (Eriksen *et al.* 1998; Scherer 2001). Lloyd (1994) found a relative use efficiency of S in cattle slurry of 55% compared to gypsum ($CaSO_4{\cdot}2H_2O$) when applied to grass intended for silage. However, this was much higher than the relative use efficiency of 5% found by Eriksen *et al.* (1995) when applying slurry to spring oilseed rape in a pot experiment. Application of mineral S fertilizer dramatically increased seed yield and S uptake of oilseed rape. In contrast, application rates equivalent to 25 and 50 tonnes of slurry per hectare did not have any effect on seed yield and only slightly increased the S uptake. So if crop yields and quality are to be maintained at present levels, the reductions in atmospheric deposition of S have to be partly replaced by use of inorganic S fertilizers. Indeed, Zhao *et al.* (2002) reviewed crop responses to S fertilization in Europe and concluded that S is now one of the most limiting nutrients for agricultural production in many European countries.

Soils most likely to show a sulphur response are free draining sandy soils with low organic matter content. Since SO_4^{2-} is readily leached from the soil there is no point in attempting to raise soil S levels by excessive fertilization. Furthermore, on livestock farms, excess S may depress the uptake of selenium in herbage and interfere with the absorption of copper by animals (Till 2002). Walker & Dawson (2002) reviewed the S fertilizer recommendations in Europe and found that most advisory bodies across Europe adopted an empirical system of advisory level of S required based on potential yield of crop, soil type and farming system.

To maintain a sufficient S supply in the future, when further reductions in the atmospheric deposition may be expected, it is important to reduce the leaching losses of SO_4^{2-}. Catch crops, especially cruciferous crops that have a high S demand and vigorous root growth, can do this effectively (Eriksen & Thorup-Kristensen 2002). In a crop rotation including both low S-demanding cereals and high S-demanding main crops (e.g. crucifers or vegetables), a suitable catch crop strategy may prevent excess SO_4^{2-} from leaching in years when low S-demanding crops are grown, and instead transfer S to the following high S-demanding crop. This is most important in low input systems, for example organic farming, but also of relevance to other farming systems. For such a strategy to work it is important that S immobilized in the catch crop is mineralized rapidly after incorporation. However, mineralization rates depend much on the choice of catch crop. The highest rates have been found for cruciferous crops and the lowest ones for legumes (Eriksen & Thorup-Kristensen 2003). Differences were partly explained by the C/S ratio of the decomposing plant material. It is generally claimed that SO_4^{2-} is mineralized from organic material when the C/S ratio is less than 200 and is immobilized if the C/S ratio is above 400, whereas C/S ratios between 200 and 400 may cause either net mineralization or net immobilization (Barrow 1960). This rule seems to apply across different organic materials such as sludge, animal manure and plant material (Tabatabai & Chae 1991; Musvoto *et al.* 2000; Reddy *et al.* 2002; Eriksen & Thorup-Kristensen 2003). Thus, it is always advisable to use supplemental S fertilizers during the incorporation of strongly immobilizing materials.

Equitable re-distribution of nutrients from manure

Intensive livestock farming usually requires purchase of supplementary feedstuff, and this represents an input to the farm which, following utilization by animals, results in a surplus of nutrients, including N and S. There are regulations aimed at minimizing nutrient accumulation in soils on animal farms, allowing a maximum animal density per unit of arable land (Sweden) or a maximum nutrient application through animal wastes per unit of land (Denmark). Nevertheless, it is almost impossible, using current manure management practices, to ensure that nutrients are equitably re-distributed. Thus the nutrient budgets of intensive livestock farms are generally imbalanced. Currently, due to the high water and relatively low nutrient contents in animal manures as compared to concentrated inorganic fertilizers, transport of these waste products is economically and practically feasible over only short distances. Furthermore, application of animal manures to soil requires more energy than harvesting operations. Economic studies indicate that the fertilizer value of solid cattle manure covers the cost of transportation only up to a distance of about 15 kilometers, and even shorter distances for slurries (Greaves *et al.* 1999).

It has been suggested that the widespread establishment of self-sustaining, mixed organic farms across regions might lead to more uniform re-distribution of nutrients present in manures, although the economic and environmental implications of such changes have not been considered. Another view is that the problem of imbalanced re-distribution of nutrients present in animal manures could be overcome, not through a restructuring of agricultural production, but by improvements in the processing of the large quantities of wastes generated by intensive livestock enterprises, so as to allow for more equitable re-distribution of the nutrients they contain.

Development of new technologies to extract nutrients out of organic wastes may provide a solution to the imbalanced re-distribution to agricultural soils. For example, one method for recovery of N and P from manures (Schuiling & Andrade 1999) is based on precipitation as struvite (magnesium ammonium phosphate). Recovery of nutrients in water-soluble forms from animal wastes would be the most desired agronomic option, providing farmers with efficient fertilizers. An optimal concept might be a centralized regional plant producing biogas from animal wastes brought from livestock farms, and including nutrient extraction and production of inorganic fertilizer compounds.

SUMMARY AND CONCLUSION

Clearly, several common critical factors need to be considered for efficient management of N and S in agricultural systems, especially for those with an emphasis on organic nutrient sources. Leaching losses of N and S, induced by additions of high quality plant residues with low C/N or C/S ratios (e.g. leguminous or cruciferous green manures) as well as large quantities of animal manures, are of major concern. Besides potentially large leaching loads, substantial emissions of N as ammonia are also associated with land-spreading of manures, whereas volatile losses of S have been much less well characterized. Large inputs of manure also increase the C content in soil and consequently enhance the potential for further losses of N via denitrification. Other volatile losses of N and S can occur during the manure handling chain, although management guidelines for storage of manures in intensive enterprises in Europe and the USA are currently well developed.

There is no doubt that organic materials represent a major nutrient input to many farming systems, particularly in the case of N from legumes and both N and S in manures. However, efficient use of the inputs is often hampered by asynchrony between plant-available release from the organic source and crop demand, primarily driven by the quantity and quality of the input. Although there are currently some options for managing these organic inputs of N and S, such as minimizing tillage, optimizing timing of manure applications and using cover or catch crops, it is critical that further innovative research is undertaken, aimed at improving the efficiency of N and S cycling in all farming systems. Approaches may include:

- selection of appropriate crop and pasture species/mixtures;
- addition of on- and off-farm wastes to manipulate soil N and S supply;
- altering the cycling of N and S via animal excreta by changing animal diets;
- developing new strategies, such as nutrient extraction from animal wastes and use of feed pad systems, to redistribute outputs from grazing animals.

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