

NITROGEN MINERALIZATION IN TEMPERATE AGRICULTURAL SOILS: PROCESSES AND MEASUREMENT

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I. INTRODUCTION

The soil nitrogen (N) cycle is an integral part of an overall, global recycling system (Fig. 1). Soils form a major repository of N within both natural and agricultural terrestrial ecosystems, containing, on a global basis, an estimated 2.4×10^{11} tons of N (Stevenson, 1982a). The soil receives N inputs through fertilizer additions and from the atmosphere in precipitation and dry deposition or via biological fixation: inputs are also made in plant and animal residues. Nitrogen is removed in the harvested crop and is lost by leaching and surface run-off of soluble forms, by gaseous transfer as nitrogen gas and nitrogen oxides (during nitrification and denitrification processes), and by ammonia volatilization. In some circumstances, erosion may also be important. In addition to these interactions with the total ecosystem, internal cycles also operate within the soil, so that even if gains and losses are in balance, then N still continues to cycle in the soil. Nitrogen is continuously assimilated into organic forms in the soil, i.e., "immobilized," and released as ammonium ions (NH_4^+) from organic matter, i.e., "mineralized." The processes of mineralization and immobilization are therefore central to the control of the flows of N within agricultural cycles and mineralization has been recognized as an important soil process since the early years of this century (Löhnis, 1910) because N in mineral forms is essential for plant growth and development.

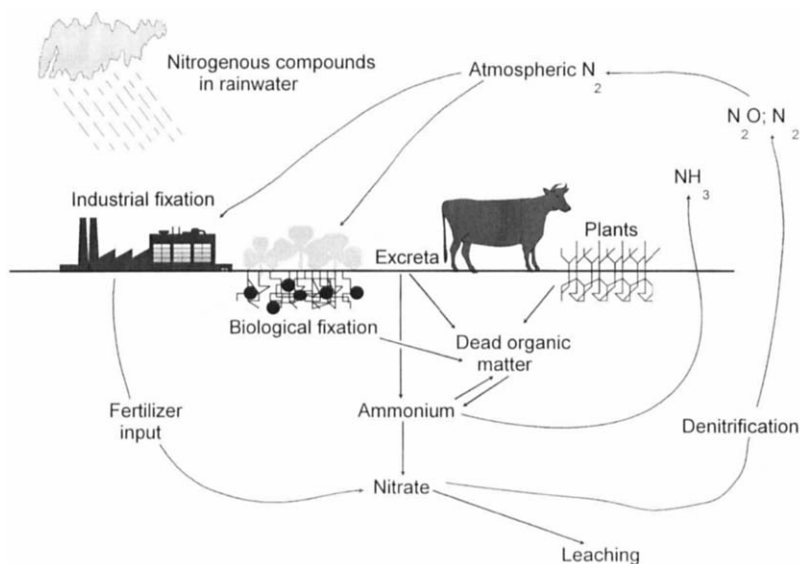


Figure 1 The "global" nitrogen cycle.

Increasingly, questions are being raised about the efficiency of N use within managed systems. Recent research has demonstrated clearly that there are substantial losses of N either to waters or to the atmosphere with potential for environmental impact. In order to improve efficiency and reduce emissions while sustaining production targets, a thorough knowledge of the N flows from all sources, especially those from organic materials, is required. Although studies of mineralization/immobilization have been undertaken over many decades, our general understanding remains fragmentary. New approaches, concepts, and information have become available which will enable a more detailed understanding of the impact of these processes on, for example, (i) improvement of fertilizer recommendations, (ii) leaching of NO_3^- (and NO_2^-) into ground or surface waters, (iii) recycling of N from crop and animal residues, (iv) production of N_2O (an important greenhouse gas) and other gaseous oxides of N, (v) changes in soil quality through changes in the nature of soil organic matter, and (vi) effective and accurate inputs to computer models to simulate N cycling.

This review therefore describes the current understanding of the conceptual basis of the processes involved in mineralization, relationships between the processes and other factors, and also how their effects can be determined practically. The aim is to present this in a way that is relevant to current and future agricultural development and to environmental issues. While we use, in the main, information that has been collected or derived from temperate agricultural systems, the general principles that we describe are applicable across other climatic zones and ecosystem types.

II. POOLS AND PROCESSES

A. SOIL ORGANIC MATTER

In undisturbed systems, soil organic matter (SOM) can be considered to attain a steady state level governed by the soil forming factors and their interaction, i.e., climate, topography, parent material, vegetation, soil flora and fauna, and time (Jenny, 1941). Where soils are disturbed and used for agricultural production, the various equilibria involved are not maintained and SOM content tends to decline. The rate of the decline and the establishment of a new steady state during cultivation will depend on soil type (and its textural, structural, and drainage status), crop rotations, and management of both residues and soil, including the intensity and mode of tillage. All of these influence the balance between inputs of residues and their breakdown by mineralization. Cropping rotations which return few residues to the soil, and do not include additions of other organic materials such as manures, show the greatest rates of decline in

organic N (Janzen, 1987), whereas the continued long-term use of N fertilizer has been shown to slow the decline of soil organic N, largely by increasing the returns of N in residues (Glendining and Powlson, 1995). However, the net retention of N from fertilizer is small; in one long-term experiment the increase in soil organic N was equivalent to only 3% of the fertilizer N applied over a period of more than a century.

In many circumstances organic matter accumulates or declines only slowly. Disturbing the soils by cultivation enhances mineralization of SOM and levels decline more rapidly. Uncultivated soils with natural vegetation tend to have most organic matter, and long-term grassland generally has more organic N than long-term arable soils. Agricultural soils therefore hold substantial but variable amounts of SOM and organic N, as shown with data taken from 106 arable fields in England (Table I). The organic N pool present in soils is always much larger than the mineral N pool; for example, mineral N measured in autumn, averaged across all of the sites in Table I for 1989–92, was only 76 kg N ha⁻¹ compared with 7 t ha⁻¹ of organic N.

SOM contains a large reservoir of nutrients and is commonly divided into a number of pools, into which materials behaving similarly are grouped. These are linked by a number of interacting, competing, and sometimes antagonistic processes. In fact, soil organic matter is composed of a continuum of materials stabilized against mineralization to varying degrees by molecular recalcitrance, physical separation from the soil microbial biomass (SMB), and/or direct asso-

Table I
The Average Topsoil Organic N Contents of 106
Arable Fields in England Sampled to 25 cm as Part
of a Long-Term Soil Mineral Nitrogen Monitoring
Scheme (ADAS, Unpublished Data)

Topsoil texture	No. of samples	Organic N (t ha ⁻¹ to 25 cm depth)	
		Mean	Range
Clay	7	11.7	8.0–13.6
Clay loam	34	7.5	5.0–12.0
Sandy clay loam	6	6.4	4.3–9.0
Silty clay loam	17	5.8	4.0–8.6
Silt loam	5	8.6	6.3–12.6
Sandy loam	31	4.7	2.7–7.6
Loamy sand	6	4.3	3.7–4.9

ciation with inorganic ions and clay surfaces. Chemical fractionation techniques have been used to define the chemical structures of SOM and have shown that a wide range of functional groups is present (Stevenson, 1982b). The relationships between these fractions and the soil N cycle have not been clearly identified or defined, and no chemical technique has yet separated SOM into biologically meaningful fractions (Paul, 1984). Four pools of SOM which made differing contributions to mineralization were identified by Paul and Juma (1981) using ^{15}N . The active pools of biomass, active nonbiomass, and metabolites were small but had rapid turnover, whereas old organic N had a much larger pool but slow turnover. A very stable pool of SOM has also been identified with a turnover rate of more than 1000 years (Hsieh, 1992), which does not take part in seasonal nutrient cycling. Physical fractionation techniques have also been used to separate SOM pools into sand, silt, and clay-sized fractions (Skjemstad *et al.*, 1988). Organic matter separated into sand-sized or light fractions declined most rapidly when soil was brought into cultivation (Dalal and Mayer, 1986), while the rate of loss of C from the clay-sized fraction was slower than that from the soil as a whole (Hassink, 1992). Physical fractionation techniques may provide a practical way to divide the SOM into fractions which can be related to the soil N cycle but this has not been explored to any great extent as yet.

Many models of SOM turnover have been developed (Jenkinson *et al.*, 1987; Parton *et al.*, 1987; Jenkinson, 1990). Bosatta and Agren (1985) introduced the idea that decomposition is a continuum, with organic matter decreasing in quality as a nutrient resource as it decays. The mathematics of this approach are complex and it is therefore not surprising that most modellers have retained the concept of a variable number of discreet pools. Contributions to those pools will be made from organic materials added to soils in crop residues after harvest or by senescence during the growing season, by incorporation of green manures or cover crops before the next planting, in waste materials (which can range from farm manures, municipal wastes, and sewage sludges to by-products from industrial processes), roots and their exudates, and from the turnover of SMB.

The simplest description of these materials is as fresh organic matter (e.g., crop residues, recent manure additions), which decomposes rapidly and is eventually stabilized to join the old/native organic matter pool, which releases N slowly (Jenkinson, 1984). The simple separation of SOM into active and passive pools has been used to model the contribution of N from recently added residues and stabilized pools (Matus and Rodriguez, 1994). More complex models separate more pools and usually include the SMB explicitly as one or more of the defined pools (van Veen and Frissel, 1981; Parton *et al.*, 1987; Hansen *et al.*, 1991; Rijetema and Kroes, 1991; Bradbury *et al.*, 1993). However, the pools defined in models are often conceptual and few can yet be measured by physical, chemical, or biological methods.

B. MINERALIZATION/IMMOBILIZATION

Decomposition and mineralization are the means by which nutrients held in organic materials in the soil are released into the soil as inorganic (often referred to as mineral) forms. Inorganic N released in this way is then available either for subsequent recycling and utilization by plants or micro-organisms or to be lost from the system. Mineralization is the transformation process whereby ammonium (NH_4^+) or ammonia (NH_3) is released by soil micro-organisms as they utilize organic N compounds as an energy source (Jansson and Persson, 1982; Royal Society, 1983). The process is complex and depends upon the activities of nonspecific heterotrophic soil micro-organisms under both aerobic and anaerobic conditions. Mineralization occurs, to differing extents, with both newly added residues and existing, already degraded organic materials of varying ages and degrees of recalcitrance (Fig 2).

Mineralization is always coupled with immobilization (Fig. 2); the two processes are intimately connected and dependent. Much of any NH_4^+ , NO_3^- , or

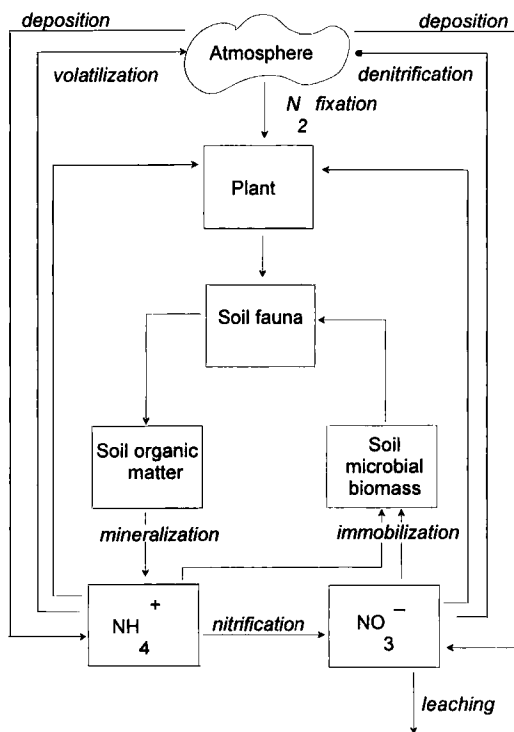


Figure 2 The soil nitrogen cycle.

simple organic N compounds that are released is assimilated rapidly by the SMB population and transformed into the organic N constituents of their cells during the oxidation of suitable C substrates, i.e., it is "immobilized." However, immobilized N is likely to be available subsequently for mineralization as the microbial population turns over. Degradation of microbial tissues is of great importance in terms of the final release of N originally bound in organic residues, and biomass N contributes, over the short term, substantial amounts of N to the pools of mobile N. Concurrent with release from the SMB will also be direct release from both fresh residues and "native" soil organic materials of various ages. Gross mineralization is the total release of NH_4^+ through microbial activities, i.e., before any immobilization back into the soil biomass. The difference between gross rates of mineralization and immobilization is net mineralization or, in some circumstances, net immobilization. Net rates are the integration of a number of soil N processes and a number of effects, which act on other interacting processes as well as directly on mineralization/immobilization: extrapolation of information to other circumstances and sites is therefore difficult.

Immobilization has been shown to occur predominantly from the NH_4^+ pool (Jansson, 1958; Recous *et al.*, 1988). However, where NH_4^+ is not available, NO_3^- is assimilated by SMB in the presence of readily available C (Azam *et al.*, 1986; Recous *et al.*, 1988). There has been no evidence of any difference between the subsequent rates of release of immobilized NH_4^+ or NO_3^- (Bjarnason, 1987). The process of continuous transfer of mineralized N into organic materials in SMB and the release of immobilized N back into inorganic pools is known as "Mineralization-Immobilization-Turnover" or MIT (Jansson and Persson, 1982). A basic assumption of MIT is that all immobilization occurs from the inorganic pool (NH_4^+ or NO_3^-). It has also been proposed that at a microsite scale there may be direct immobilization of small organic compounds such as amino acids (Hadas *et al.*, 1987; Drury *et al.*, 1991), known as the "Direct Hypothesis." Although it has been demonstrated that SMB can utilize amino acids in this way (Barak *et al.*, 1990), MIT generally describes overall mineralization more accurately (Barak *et al.*, 1990; Hadas *et al.*, 1992). However, since both mechanisms are not mutually exclusive and both have been demonstrated, both may occur concurrently, at least in some circumstances.

The way that the mineralization/immobilization processes operate is of importance to the turnover, recycling and fate of released N and many questions remain. For example, we do not know how much competition occurs between plant roots and loss processes and immobilization for the products of gross mineralization or whether SMB absorbs all the N that it requires so that the remainder is available for plant uptake or loss. Similarly, the microsite distribution of net mineralized N and how this relates to interpretation of measurements of soil mineral N measurements is not known. In a practical sense, it is the balance between mineralization and immobilization (i.e., net mineralization)

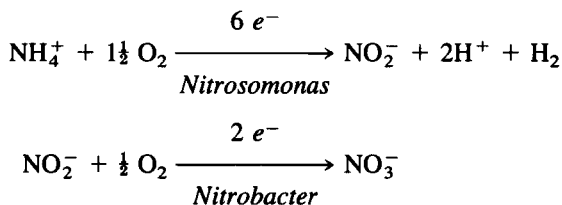
which seems to be important in influencing N available for crop uptake or loss to waters or the atmosphere, and it is this which is most often measured or estimated. However, gross mineralization is the more fundamental process which reflects the properties of the substrate and its interaction with the ambient environment. An understanding of gross mineralization will provide the sounder basis on which to base mechanistic models or allow confident extrapolation. Gross mineralization is more difficult to measure but new techniques and approaches have become available over recent years (see Section IV.E.).

Immobilization is involved with all mineralization activities as part of an intricate cycle of organic matter degradation and renewal, and the assimilation of mineral nutrients to provide the basis for the multiplication and maintenance of the SMB. In theory, the balance between mineralization and immobilization can fluctuate from positive to negative according to ambient soil and environmental conditions and the quantity and quality of available organic substrates. In practice, in most situations, there is usually net mineralization during an annual cycle, although over the shorter term immobilization may dominate, for example when organic materials such as cereal straw are returned to the soil (Ocio *et al.*, 1991).

Long-term accumulation of organic N in soils may also be referred to as "immobilization." This is not a direct result of SMB activities, but represents the N returned to the soil in plant shoot and root residues. It is quite usual and reasonable for a soil to have net positive mineralization and to release mineral N, but to accumulate organic N during the year, provided that the N input was greater than any offtake in harvested crops plus that accumulated in organic matter. This is particularly the case in undisturbed grassland soils where there is usually long-term accumulation of organic N through this means.

C. NITRIFICATION

As well as the linkage between mineralization and immobilization, it is also important to consider interactions with nitrification. In most circumstances nitrification is the oxidation of reduced N compounds, primarily NH_4^+ , by two groups (in the main) of autotrophic bacteria (*Nitrosomonas* and *Nitrobacter*) according to the following pathways.



The transformation from a relatively immobile (NH_4^+) to a highly mobile state (NO_3^-) via nitrite (NO_2^-), provides a key step in the soil N cycle often leading to an excess of NO_3^- and subsequent N loss by leaching or denitrification. In many circumstances the major source of NH_4^+ is through release by mineralization; exceptions are addition of NH_4^+ or urea-based fertilizers, or where animal excreta are being returned. Despite its importance as a rate-limiting process controlling the availability and loss of NO_3^- , and despite a reasonable knowledge of the ecology of the bacteria involved (Prosser, 1986), nitrification is poorly defined in many soils. In arable and other tillage systems, it is usually assumed that nitrification is not a limiting process. Conditions are such that rapid oxidation takes place and it is unusual to find accumulations of NH_4^+ , i.e., nitrification rate exceeds that of mineralization. However, the situation differs for grassland soils especially where swards are grazed or farm wastes are applied and there are large returns of ammoniacal N. Under these circumstances there may be significant quantities of NH_4^+ in the soil profile at various stages through the year (Jarvis and Barraclough, 1991).

Nitrification is dependent on soil aerobicity (and thus soil texture, structure, and water contents), on pH (i.e., inhibited by high pH caused by liquid or anhydrous ammonia fertilizer, for example), and on substrate (NH_4^+) availability and appropriate populations of micro-organisms. Nitrification interacts strongly with ambient local soil conditions and there is a high degree of spatial compartmentalization of NH_4^+ production and consumption sites. This, coupled with diffusional constraints between microsites, controls the rate at which nitrification proceeds (Bramley and White, 1991).

In grassland soils there are known to be latent nitrification potentials which have developed because of previous conditions and managements and which can be displayed over both the relatively short (Jarvis and Barraclough, 1991) and the longer term (Willison and Anderson, 1991). Nitrification rates are also strongly influenced by additions of mineral N as fertilizers. At low inputs of NH_4^+ to grassland soils it has been suggested that competition between plant uptake and nitrification reduces nitrification rates (Barraclough and Smith, 1987). Where there was considerable excess of input over removal by plants, nitrification rates were substantially higher.

It is worth emphasizing that nitrification is central to the flows, losses, or utilization of N through the conversion of NH_4^+ into labile NO_3^- . Its greater understanding is an important further step in maximizing N efficiency and reducing losses, particularly in grassland, but also as a key interactive process linked by flows of substrates and spatial distributions in all ecosystems.

D. SOIL BIOMASS

Biomass is at the center of the internal soil N cycle (Jenkinson, 1990) and is important as (i) an agent of change, decomposition, and release of N (and other

nutrients) from fresh organic residues and native SOM into more labile forms, (ii) a major sink for "active" soil N, and (iii) a potential source of labile N. Biomass N is in a constant state of turnover (Jenkinson and Ladd, 1981) and represents a significant proportion of the total soil N which remains relatively constant throughout the year (Holmes, 1994). For example, SMB N alone accounted for 0.5–15.3% (3–108 kg N ha⁻¹) of the total N in the surface 12.5 cm of the soil (Anderson and Domsch, 1990), and Jenkinson and Ladd (1981) showed that in an unmanured wheat field (to 23 cm depth) SMB contained 95 kg N ha⁻¹. The annual N flux through the SMB can be greater than the offtake in harvested grass (Brookes *et al.*, 1984). Bristow and Jarvis (1991) have shown that, in cut and rotationally grazed pastures with a range of N inputs, the average N content of the SMB (0–10 cm depth) ranged between 138 and 240 kg N ha⁻¹. The turnover time for N derived from microbial biomass has been estimated to be from 5 (Marumoto *et al.*, 1982a) to 10 (Smith and Paul, 1990) times faster than that in SOM or returned in plant residues. This rate of transformation is assumed to be very important in those production systems with low or no annual inputs from fertilizer or other immediately available sources. However, the exact nature of the participating populations, the community structure, and the prediction of their ability to react to environmental and other changes requires characterization and definition. Understanding these issues will be a key to future successful management of low-input systems.

The soil biomass comprises almost every class and order of invertebrates, as well as a wide range of fungal and bacterial species and genotypes. The community structure and size have been shown to be related to soil type and management (Chaussod *et al.*, 1988). Thus, mesofauna activity decreased with intensity of management (Anderson, 1988) and cultivation has resulted in high earthworm mortality (Curry and Byrne, 1992). Heterotrophic microflora are the primary decomposers and contribute more than 90% of the energy flux in soil (Ladd and Foster, 1988). They comprise diverse assemblies of organisms which are often divided into two groups, i.e., autochthonous species which are able to maintain low and relatively constant activities in utilizing the more resistant SOM, and zymogenous species which react rapidly to inputs of readily metabolized substrates but then return to dormancy (Ladd and Foster, 1988). The existence of these two types has been used to explain changing C and N dynamics during decomposition of straw (Cochran *et al.*, 1988) and the occurrence of a greater proportion of active biomass in the upper layers of grassland soils (Hassink, 1993). Although management has been shown to have an effect on SMB activities and possibly on community structures (Lovell *et al.*, 1995), substantial differences in management of grassland soils have had little or no effect on overall biomass size (Bristow and Jarvis, 1991; Lovell *et al.*, 1995). Resolution of SMB into labile and resistant components has allowed the development of a recent model to simulate changes in active vs quiescent biomass fractions (Grant *et al.*, 1993). The fungal:bacteria biomass ratio has been shown to be an impor-

tant factor regulating the relationship between activity and SMB size (Sakamoto and Oba, 1994). Techniques involving antibiotics have been developed to separate effects of these two groups of organisms (Landi *et al.*, 1993).

Mineralization should not be considered totally as a microbial activity (Woods *et al.*, 1982). Invertebrate fauna make an important contribution by (i) redistributing organic materials over a range of spatial scales, (ii) enhancing the rate of cycling through chemical change during metabolism, and (iii) having an effect on microbial populations themselves by creating or removing appropriate conditions for their various activities. Soil invertebrates thus contribute to N fluxes by changing microsite environments and controlling populations of other organisms, and through trophic transfers in food webs and turnover of tissues (Anderson, 1988). Although there is no clear consensus about the net effects of feeding and other activities of micro-, meso-, or macrofauna, it is, however, clear that they have substantial impact. Thus earthworms have increased CO₂ evolution, decreased SMB, and increased mineralization (Ruz Jerez *et al.*, 1988), mineral N levels were higher in casts than in surrounding soil (Scheu, 1987) and denitrification has been increased (Elliott *et al.*, 1990). The main impact of earthworms, however, was to bring plant residues and soil materials into close contact (Curry and Byrne, 1992).

The role of microbivorous fauna (e.g., protozoa, nematodes) has been extensively studied (Bouwman *et al.*, 1994) and there have been suggestions that predation stimulated N turnover since more N was mineralized when protozoa were present (Woods *et al.*, 1982; Kuikman and van Veen, 1989). Recent studies (Yeates *et al.*, 1993) have shown the importance of microfaunal grazing on microfloral populations and nutrient cycling. However, Hassink *et al.* (1993) did not find any effect of grazing on microbial activity. It is also thought that bacterial predators can enhance efficient recycling of N in root exudates (Griffiths and Robinson, 1992; Griffiths, 1994).

It is clear that interactions within the soil biomass and between this biological component of the soil and the net effects of mineralization and other N cycling processes are complex and poorly understood but of much importance since key functions in many biologically based soil processes are involved (Powlson, 1994; Patra, 1994). The influences on net release of mineral N are profound but it is not yet possible to manipulate N flows through these complex food webs to the advantage of management and environmental impact. Interactions of this nature may be more significant in relatively undisturbed systems such as grassland or in low-input and organic managements, but an understanding is essential to aid the efficiency of N utilization in all systems.

E. EFFECTS OF ORGANIC MATTER ADDITIONS

Recent additions of organic materials have the potential to mineralize at the greatest rates. Shen *et al.* (1989) showed that fresh residues were about seven

times more decomposable than SOM in arable soils. Fresh materials may be returned to the soil either gradually in small amounts throughout the growing season, e.g., by leaf-drop or root exudation, or in larger pulses, e.g., after incorporation of a cover crop or harvest residue, or application of slurry or manure.

Roots and root exudates offer a source of organic matter for mineralization during the growing season for both arable and grass crops. A larger pulse is made available when the soil is cultivated. Below ground, plant biomass holds substantial amounts of N (Saffigna, 1988), the rate of breakdown of which will depend on composition. It has been estimated that between 10 and 40% of the increase in root dry matter is exuded from the zone immediately behind the root tip (P. Darrah, personal communication), which provides substrates for hotspots of microbiological activity in the rhizosphere. The effects of this activity may be beneficial, harmful, or neutral for crop growth depending on whether net mineralization or net immobilization occurs. Robinson *et al.* (1989) showed that the potential increase in N availability to the roots was greater where exudates had a high C:N ratio. However, Griffiths and Robinson (1992) showed that the main effect of root exudation was to allow recycling of N in the exudates in forms available to plants, rather than causing any extra release from SOM.

The amounts of N returned to the soil following crop harvest can vary widely depending both on crop quality and yield and on the residue management strategy (Table II). Large amounts of N, in excess of 200 kg N ha⁻¹, can result after the harvest of vegetable crops (Rahn *et al.*, 1992). The use of cover crops or green manures may return from 10 to 150 kg N ha⁻¹ (Christian *et al.*, 1992) when residues are incorporated as young and fresh materials shortly before drilling the next crop. Some cover crops, e.g., mustard, are affected by frost so that residues resulting from defoliation occur earlier (Shepherd, 1992). Other crops are grazed and large amounts of N are returned in feces and urine. The effects of all these returns are considerable and have been reviewed recently (Shepherd *et al.*, 1996).

Even in grassland after either cutting or grazing, much plant material remains in the stubble and roots and there is a continuous turnover of N through the leaf litter and roots (Parsons *et al.*, 1991). The quantities of roots and macro-organic matter in the soil increase with the age of the sward (Garwood, 1967). Where swards are cultivated and resown or brought into arable production, the macro-organic fraction has been shown to be relatively labile (Warren and Whitehead, 1988) and large amounts of N may be released for a number of years. Potential mineralization has also been related to the "light" SOM fraction in a sandy soil (Gaiser and Stahr, 1994). Francis *et al.* (1992) measured 230 kg N ha⁻¹ accumulated in the stubble and roots of a 3-year ryegrass/clover sward. Whitehead *et al.* (1990) measured 536 and 602 kg N ha⁻¹ accumulated in stubble, leaf litter, roots, and macro-organic matter in an 8- and 15-year-old sward, respectively.

Table II

Typical Balances between N Uptake (or Fixation) and N Removed in the Harvested Produce of Arable Crops in the United Kingdom on >100,000 ha (Sylvester-Bradley, 1993)

Crop	Yield (t ha ⁻¹)	N content (kg t ⁻¹)	N uptake ^a (kg ha ⁻¹)	N offtake ^b (kg ha ⁻¹)	N left ^c (kg ha ⁻¹)
Winter wheat (milling)	7.5	19	190	140	50
Winter barley (feed)	6.5	17	150	110	40
Spring barley (malting)	4.5	14	80	65	25
Winter oilseed rape	3.2	32	250	100	145
Sugar beet	42.0	1.7	200	70	130
Maincrop potatoes	40.0	2.5	200	100	100
Linseed	1.8	38	90	70	20
Winter field beans	3.5	42	225	145	80
Spring oats	4.7	17	115	80	35

^a Yield × N content of whole above-ground crop.

^b N removed in harvested crop.

^c N remaining in crop residue after harvest.

Young (1986) calculated that N mineralization increased after ploughing from approximately 100 kg N ha⁻¹ for a 1-year ley to 280–380 kg N ha⁻¹ for leys older than 4 years. It has been shown that grassland management, e.g., N inputs and grazing, influences N accumulation and, hence, mineralization after ploughing (Whitehead *et al.*, 1990). Shepherd (1993) found that soil mineral N contents were of no value in predicting response of wheat to N after ploughing grass swards because subsequent immobilization/mineralization processes were too variable.

Biologically fixed N may also increase the N content of specific crop residues and, thus, potential mineralization rates. Further, there has been some evidence that white clover (*Trifolium repens* L.) has an effect in increasing mineralization perhaps because of high N contents in roots and nodules, but also because soil structure may be improved in the presence of the clover which could increase mineralization (Mytton *et al.*, 1993). The effect of clover is uncertain and the clover content of sward and its contribution to N cycling have only rarely been well defined.

Animal manures are often added in large quantities to soils and are a mixture of NH₄⁺, urea, uric acid (especially in poultry manures), and more complex materials such as proteins, either undigested or partially digested during transit through the gut (Chescheir *et al.*, 1986). Manures vary in both total N content and the forms of N present which are affected by many factors, e.g., animal species, diet, amount and type of litter, and dilution by water. The main source of carbon in animal slurries or layer poultry manures will be that which has passed

through the animal. However, bedding will introduce other carbon sources and the interaction of manure and bedding, either during storage or after application to the land, will have important effects on the release of N from organic forms and subsequent nitrification (Jensen, 1931), but this is poorly defined. Dung and urine directly voided to pastures by grazing animals behave differently to stored manures (but have enormous effects on N transformations) (Jarvis *et al.*, 1995). Storage of manures before application to land results in some losses of N and other nutrients, and changes in the pools of residual N. Composting of manure caused a large reduction in volume and produced a material with an increased proportion of stable organic materials with a higher C:N ratio (Lampkin, 1990). Anaerobic storage of manures or slurries results in materials with high NH_4^+ contents. Thus, on average, 23% of the N in dairy cattle manure was lost as NH_3 during composting (Lampkin, 1990), compared to 10–15% during anaerobic storage. However, losses after application to land are much increased in manures which have been stored anaerobically, since they contain a much greater proportion of NH_4^+ (Pain *et al.*, 1993). The losses of N by volatilization during composting have been reduced by increasing the initial C:N of the compost (Kirchmann, 1985) and where compost heaps are not covered, then substantial losses of potassium (K) and N as NO_3^- may occur through leaching (Russell, 1973).

Since urea and uric acid hydrolyze rapidly to NH_4^+ after application, these compounds are usually included as part of the inorganic N pool of manures and the hydrolysis is not considered strictly as mineralization. The amount of N released in available forms from manures has often been calculated assuming that the inorganic N (NH_4^+ , urea, and uric acid) and organic N pools supply N independently, e.g., $\text{N supply} = E_1 N_i + E_2 M N_o$, where N_i = inorganic N content, N_o = organic N content, E_1 and E_2 are efficiency factors for N which allow for loss processes, and M is the proportion of organic N mineralized (Sims, 1986). However, stimulation of biological activity in soils due to the readily available C added in manures can lead to rapid and significant immobilization of the manure inorganic N pool (Flowers and Arnold, 1983; Bernal and Kirchmann, 1992). The interactions between the N pools in manure and soil biomass and the inherent variability of manures make the prediction of the timing and amount of N release from manures difficult. There is a need for better characterization of the forms and amounts of N in animal manures and better assessment of the availability and interactions between these pools under field conditions. Effects of cattle slurry on N release have been found to at least 0.5 m depth when applied to grassland (Kandeler *et al.*, 1994).

Sewage sludges may be applied to agricultural land when they do not contain significant amounts of metals or other toxic chemicals and can provide a significant input of organic materials. As with animal manures, sewage sludges contain both inorganic and organic N fractions. The relative proportions of inorganic and organic N in sewage sludge and the composition of the organic fraction (i.e., its

recalcitrance) depend on the sewage production process and effects are therefore variable (Smith *et al.*, 1992).

III. PROCESS CONTROLS

Mineralization is a highly variable process and although there have been many studies over several decades, predictive capability is poor. The rate at which mineralization occurs is the result of complex interactions between biological, chemical, and physical components of the soil and is subject to many external influences that have been reviewed extensively (e.g., Haynes, 1986). Basic knowledge was established about mineralization kinetics in the early 1970s, mostly derived from experiments under controlled environmental conditions and a first assumption (Stanford and Smith, 1972) was that mineralization depended upon first-order kinetics with only one pool of degradable SOM that was available for utilization and N release. Because the concept of multiple organic-N pools (Section II.A) with different decomposition rates has been developed recently (Bonde and Lindberg, 1988; Hassink *et al.*, 1990; van Veen *et al.*, 1985; Warren, 1985), it seems likely that mixed-order kinetics will provide a better description. More usually, multiple pools are considered in models, each having a different rate constant for decomposition, but with first-order kinetics.

A. RESOURCE QUALITY

Soil organic matter is a very variable material and, in the first instance, is influenced by the nature of the returned organic materials which has a substantial effect in determining mineralization. Russell (1973) described crop residues as comprising three separate groups of materials, (i) cell wall and structural materials, consisting of the skeletal framework (i.e., cellulose) and cementing/encrusting materials (with carbohydrates predominating in young shoots and lignin accumulating in older tissues), (ii) reserve substrates including starches, fats, and proteins, and (iii) cell contents (i.e., proteins, sugars, unassimilated NO_3^- , and traces of NO_2^- and NH_4^+). Mengel and Kirkby (1978) estimated that the relative proportions of N components in residues were as follows: inorganic fraction (NO_3^- , NH_4^+ , NO_2^-) generally <2%; soluble amino compounds (acids, amides, and amines) approximately 5%; and protein and nucleic acids 90–95%. All three fractions are influenced by crop growth and nutrition, especially the supply of N. The N concentration in plants declines with age because a greater proportion of resource is diverted to the production of N-free material required for structural support (Raymond *et al.*, 1960). Increasing the N supply increases

all fractions, but not to the same extent, and the largest effect is on soluble amino compounds. Once in the soil, the simpler N compounds mineralize more quickly than complex materials (Rubins and Bear, 1942). Lignin, for example, is resilient to attack and it may also protect cellulose by encrustation, i.e., an example of physical protection and chemical composition influencing mineralization. The relative proportions of the different N forms in crop residues have a marked effect on the rate at which breakdown occurs (Waksman and Tenney, 1927).

The nature of the returned organic materials is therefore of some importance; C:N characteristics of materials provide an indication of the likely balance between mineralization and immobilization processes when residues are added to soil. However, there is little evidence to suggest that critical C:N ratios for either SOM or added materials can be defined which are generally applicable to aid prediction of mineralization rates (Haynes, 1986). In many respects this would be useful in providing an easily determined characteristic, but the wide range of other constituents (e.g., lignin and polyphenol contents) and their behavior makes dependence on interpretation of C:N too simplistic.

C:N status does, however, provide an important guide to whether net N mineralization or immobilization will occur when organic materials are added to soil. In general, as substrate C:N increases, mineralization decreases until some critical point is reached (Marstorp and Kirchmann, 1991). This relationship has not always held (Douglas and Magdoff, 1991) and has been criticized since not all the C and N is readily available to micro-organisms (Frankenberger and Abdelmagid, 1985; Reinersten *et al.*, 1984). The chemical composition, competitive activities, and community structure of the SMB also vary between different systems and therefore influence the efficiency of C use and N demand (Ladd and Foster, 1988). This in turn influences the reliability of simple ratios as an index of mineralization. Attempts to provide a more mechanistic approach by considering C either alone (Douglas and Magdoff, 1991) or in conjunction with specific components such as lignin (Frankenberger and Abdelmagid, 1985) have met with only limited success. Jenkinson (1984) suggested that the C:N ratio was a good guide to the amount of N released from a residue in a year in temperate arable soils with typical crop residues. However, to make progress it will be important to define differences in the short-term release patterns for different residues with widely different characteristics as illustrated in Table III.

The C:N of manures has also been suggested to have a major influence on subsequent mineralization in soils. Salter and Schollenberger (1939) found that yield depressive effects were not serious unless the manure C:N ratio exceeded 20, whereas Kirchmann (1985) suggested that a C:N of 15 was critical, with N immobilization occurring above and mineralization below this value. This critical ratio of 15 has further been corroborated by Castellanos and Pratt (1981) and Beauchamp (1986). Analysis of 36 cattle farmyard manure samples found an average ratio of 14, with 14 samples having a ratio > 15, i.e., with a potential for

Table III

Examples of C:N Ratios of Arable Crop Residues and N Return (kg N ha^{-1})
(Source: ADAS, Unpublished Data, Unless Otherwise Stated)

Crop	Residue	No. of samples	C:N ratio			N return in residues
			Mean	S.D.	Range	
Sugar beet	Tops	35	17.1	2.35	12.1–25.0	118
Potatoes	Haulm ^a	10	22.7	5.41	15.5–30.3	41
Dried peas	Haulm ^a	20	36.9	12.70	19.5–62.5	48
Field beans	Haulm ^a	14	43.6	6.87	32.8–56.3	45
Oilseed rape	Haulm ^a	—	55.0	—	22–74 ^b	87

^a Sampled at harvest, and therefore does not include material senescing before harvest.

^b From Holmes (1980).

net immobilization (ADAS, unpublished data). Castellanos and Pratt (1981) reported C:N values of 6.5 for chicken manure and 9.9–15.9 for cattle and pig manures. In a UK survey (ADAS, unpublished data), average values for cattle and pig slurry were 5 and 2, respectively. However, as large proportions of the N in all the above materials were NH_4^+ , it is more meaningful to look at the solid fraction alone where typical ratios were 13 and 7 for cattle and pig slurries, respectively.

B. ENVIRONMENTAL CONTROLS

Availability of appropriate sites with appropriate conditions for mineralization, and any other transformation, is controlled by the relationships between the soil and the external environment, especially those factors which influence the degree of soil aerobicity. Aeration status is dependent upon soil texture, structure and moisture, the cultural management of the system and the returns of organic materials. Soil temperature and moisture then further interact by influencing microbial activity. Physical location of organic materials at a microscale is therefore of some importance: distribution in locations with access to O_2 increases the potential for N release. Significant proportions of SOM are associated intimately with clay and other inorganic components of the soil matrix which restricts the potential for mineralization. The vertical distribution of organic materials in the soil profile also has effects. Thus, although the concentration of mineralizable N has been shown to decrease exponentially with depth (Power, 1980), laboratory incubations of soils from different depths have indicated that although the majority of mineralization occurs in the top 20 cm, 20–30% of total N released would

be mineralized below 60 cm (Cassman and Munns, 1980; Hadas *et al.*, 1986). Mineralization in the subsoil may therefore contribute substantially to the amount of N mineralized, and this has important implications for leaching losses.

Mineralization is influenced profoundly by the temperature changes that are normally encountered under field conditions, with a lower limit close to freezing. The majority of soil micro-organisms are mesophyllic and prefer moderate temperatures within optimum activity between 25 and 37°C and a base temperature of 5°C. Freezing/thawing changes may have comparable effects to those of wetting/drying and result in the release of soluble materials and/or disruption of soil aggregates. As well as mesophiles, psychrophiles, which are active at lower temperatures, are also common and have been shown to play an important role in British soils (Dickinson, 1974). This has implications, therefore, for release of N over winter during periods of maximum drainage. Recent studies in long-term grassland soils have indicated that between 21 and 38% of a total net annual mineralization occurred during the period November–February (Gill *et al.*, 1995): this again has implications for leaching losses. In general, net mineralization rates increase with temperature and tend to become less variable at higher temperatures (Stanford *et al.*, 1973). At optimum soil water content, an Arrhenius function with a Q_{10} of approximately 2 described the relationship between temperature and net N mineralization (Kladivko and Keeney, 1987).

Changes in the availability of soil water have a number of effects: (i) deficiency/stress limits biological activities and hence mineralization; (ii) excess reduces aerobicity and therefore alters the activities of different microbial populations in different microsites, i.e., reduces mineralization; (iii) soil water content controls solute diffusion and mass distribution of the products of microbial activity; and (iv) cycles of wetting/drying increase the availability of substrates (Cabrera, 1993). Drainage of permanent grassland soils increased the amounts of NO_3^- leached because 'it was thought' of an enhanced mineralization capacity where aerobicity was increased (Scholefield *et al.*, 1993). Some bacteria can operate under anaerobic conditions but their rate of activity is much lower than under aerobic conditions. The mechanisms involved during wetting and drying are not clear, but may involve a release of energy supplies after physical disruption and/or increased accessibility of degradable organic materials and, in the first instance, a significant flush of mineral N through a release of readily mineralized materials from freshly killed biomass cells (Marumoto *et al.*, 1982b).

Moisture distribution in the soil appears to be more important than the absolute amounts present (Cassman and Munns, 1980). Optimum net mineralization rates have been achieved at between -0.33 and -0.1 bar where water occupied 80–90% of the pore space; the rates fell as soil moisture potential fell below -0.33 bar (Stanford and Epstein, 1974). At very high soil moisture contents the mechanisms which control mineralization are unclear and both increased and decreased rates of mineralization have been observed. Although moisture/temperature re-

gimes are likely to be the most important external controls over mineralization rates, there is contradictory evidence for a significant interaction between the two variables (Cassman and Munns, 1980; Kladvko and Keeney, 1987). It has been suggested that separate equations to describe moisture and temperature effects can be used to predict mineralization rates without introducing significant errors (Kladvko and Keeney, 1987). However, whereas net mineralization rates in grassland soils throughout the year were not significantly influenced by variation in soil moisture, between 31 and 41% of the variance was accounted for by soil temperature (Gill *et al.*, 1995).

There have been suggestions that adding immediately available N, in fertilizers for example, stimulates mineralization, a so-called priming effect. However, it has been shown by both modelling (Jenkinson *et al.*, 1985) and experimental studies (Hart *et al.*, 1986) that this is usually an artifact of procedure and unlikely to be significant in most circumstances.

Soil texture exerts an important control over mineralization by (i) influencing aeration/moisture status, (ii) affecting the physical distribution of organic materials and hence their potential for degradation, and (iii) conferring some degree of "protection" through an association of organic materials with clay particles (Hassink *et al.*, 1993). Dickinson (1974) suggested that adsorption of micro-organisms on the surface of clays and the creation of protected sites was an important controlling factor whose impact increased as soil clay content increased. Other studies have shown (Gregorich *et al.*, 1991) that there was a close interaction between SMB and their decay products in clay soils which allowed a more efficient transfer of nutrients between subsequent microbial generations. Major differences in mineralization rates in soils of different textures have been found and the fraction of small pores, $<1.2 \mu\text{m}$, explained more than 50% of the variation in N mineralization rate between soils (Hassink, 1992). Over 70% of the variation was explained by the soil particle size fraction $<50 \mu\text{m}$ (i.e., silt plus clay content). Thus it seems probable that a major proportion of SOM that is vulnerable to mineralization may be located in pores or on surfaces which cannot be reached by micro-organisms or do not have the requisite degree of aerobicity and therefore can be regarded as protected. This therefore occurs to a greater extent in finer than in coarser textured soils resulting in lower net mineralization rates (Verberne *et al.*, 1990). The degree of structure development and the relationship between particles and pores are therefore extremely important and have implications for understanding the effects of cultivation on mineralization (Hassink, 1994). Physical disruption of the soil increases both aeration potential and microbial access to readily mineralizable SOM (Cabrera and Kissel, 1988) and provides at least a partial explanation for the flush of mineral N that is released when grassland is ploughed (see Section III.C).

Other soil factors are also important and include pH [decomposition is inhibited under strongly acidic conditions; Jenkinson (1981)] and contamination of

soils by, for example, heavy metals which have immediate impact in reducing mineralization rates (Chang and Broadbent, 1982). The presence of plants can be expected to influence mineralization rates through an increase in the availability of substrates. The importance of this depends on the C:N ratio of the materials either exuded from the roots or returned to the soil during senescence (Section III.A). After five species of grasses were grown in the same soil for 3 years (Wedin and Tilman, 1990) there were 10-fold differences in net mineralization which corresponded to species differences in tissue N concentration, below-ground biomass (roots), and lignin contents.

C. EFFECTS OF CULTIVATION

Cultivation, its timing and methodology, is an important tool for modifying mineralization of old organic matter as well as recent additions. Mechanical disruption of the soil structure makes previously protected SOM available for degradation and increased rates of mineralization have been observed in disturbed soils (Ballesdent *et al.*, 1990). The cultivation method, type of machinery used, and the energy input into a soil all have effects on the disruption of soil structure and therefore can be expected to affect the amount of N mineralized. Ploughing has caused more mineralization than establishing crops by direct drilling. Goss *et al.* (1993) estimated that extra net mineralization of approximately 20 kg N ha⁻¹ occurred with ploughing as compared with direct drilling, although other studies have shown smaller differences (e.g., Powlson, 1980). Where residues were left on the soil surface, mineralization rates were reduced and immobilization rates enhanced, compared with an incorporation management (Power *et al.*, 1984). Richter *et al.* (1989) showed that deeper ploughing slowed mineralization, but Francis *et al.* (1992) in contrast found no differences between mouldboard and chisel ploughing a grass ley. Similarly, Webb *et al.* (1991) and Lloyd (1992) found only small or no differences between ploughing compared with direct drilling or minimum cultivations following a grass ley. Timing of cultivation also affects mineralization patterns and, therefore, availability of NO₃⁻ for crop uptake, leaching, or other processes. Vinten *et al.* (1992) and Stokes *et al.* (1992) have shown that delaying cultivations in autumn significantly reduced NO₃⁻ release.

It is also important to consider the effect of adverse soil conditions on mineralization. Compaction, for example caused by heavy machinery or cultivating the soil at the wrong time, reduces soil porosity, which modifies and reduces net mineralization. Furthermore, the age and dynamics of SOM in relation to the soil structural hierarchy (Dexter, 1988), which will be determined in large part by cultural procedures, are thought to be important in separating SOM into pools which are biologically significant. Correlations have also been established be-

tween the bacterial component of SMB and a specific pore size fraction (Hassink *et al.*, 1993).

D. MICROSITES, DIFFUSIONAL CONSTRAINTS, AND SOIL ARCHITECTURE

When inorganic N is extracted from soil it represents the sum of that in many microsites. Although the bulk concentration of NH_4^+ is not thought to control microbial processes such as immobilization and nitrification, their rates may vary widely even when the bulk soil NH_4^+ is constant (Davidson and Hackler, 1994) through microsite heterogeneity. There are substantial diffusional constraints to the movement of ions in soils and these result in a diversity of micro-organisms within close spatial and temporal proximity which otherwise which would not occur (Focht, 1992) and which influence further transformation and transfer of N. Restricted diffusion of NH_4^+ may result in preferential immobilization by SMB (Davidson *et al.*, 1990) and only the N in excess of microbial requirements enters the available NH_4^+ pool (Drury *et al.*, 1991). Where organic N resources are low, little NH_4^+ will be released and any mineral N diffusing to a particular site will be immobilized. Recent studies of soil N cycling, and mineralization in particular, have considered the architecture of the soil in relation to the internal N cycle and the distribution of soil organisms (Drury *et al.*, 1991; Killham *et al.*, 1993). Any such consideration has to be made in association with a knowledge of the diffusion constraints to NH_4^+ and NO_3^- movement between microsites, i.e., in different physical pools (Darrah *et al.*, 1983). On this basis, a mechanistic understanding of the controls over mineralization at a fundamental level can be developed and it should then be possible to integrate effects sequentially into a series of larger scales, taking into account the unique spatial and temporal properties of each one. A model of soil N cycling based on more realistic, diffusion-limited activities should provide a more fundamental understanding of mineralization and perhaps allow, with other appropriate data, direct interpretation of soil mineral N contents. The concepts of microsites and different "pools" of potentially mineralizable N and mineral N are implicit in this approach.

Much is also known qualitatively about many of the soil factors affecting mineralization (Section III). While there is quantitative information on temperature and moisture, this is restricted and does not address interactions with other factors and therefore does not, at present, allow a practical interpretation of the impact of mineralization and recycling other than in a few specific circumstances. Further, the current information does not allow the development of effective broad-brush predictive models. Progress requires better definition not only of the SOM, but also its location and interaction with soil biology and environmental status. The relationships between pore size (resulting from an

interaction between soil texture and structure) and mineralization, and whether "protected organic matter" is of practical relevance, need to be determined.

IV. MEASUREMENT AND PREDICTION OF MINERALIZATION

A. BACKGROUND

Methods used to measure mineralization and predict its relevance for plant uptake, losses to the environment, and overall N cycles have, for the most part, relied on determination of net mineralization, i.e., the balance over immobilization. Many methods to assess net mineralization have been described, but because it has been seen to be of only limited immediate practical use, and because it has been difficult to measure, gross rates of mineralization have rarely been determined. The recent development of stable isotope methodology has meant that investigation of gross mineralization is now more easily achieved and is an important development to aid further understanding.

It is possible to partition the available methods to assess mineralization in various groupings according to different criteria. In this review we identify three main groupings, i.e., those which (i) rely on either the plant as a means of estimation or a knowledge of N budgets for the system, (ii) attempt to provide indices of potentially available N by chemical extraction or by incubation under controlled conditions, or (iii) aim to determine, either directly in the soil profile or under conditions as close as possible to those in the field, the products of the processes involved, i.e., NH_4^+ and NO_3^- . The aim of all these methods has been to provide a more precise basis for evaluating the capacity of soil to supply N. We discuss the more important techniques that are currently used and indicate their potential for further use as well as their shortcomings.

B. MINERALIZATION FROM N BALANCE/CROPPING DATA

One of the simplest methods to provide information on the supplies of N from native SOM sources is to use the crop as a sink for any N released, i.e., by growing plants under natural conditions in the absence of external inputs (fertilizers, excreta, fixation, and, ideally but not often easily achieved, atmospheric deposition). Uptake and removal into the crop are then assumed to equate with mineralized N; such information is an essential component of many fertilizer response trials and thus many data sets exist from which mineralization can be estimated. While this can provide useful information it does require careful interpretation in order to be translated into net mineralization rates because (i) N released and partitioned into nonharvested components, i.e., roots, stubble, senesced leaves, etc., is not accounted for, (ii) there may be significant losses

through leaching or denitrification, (iii) there may be significant influence of the plant itself through the release of exudates and their impact on soil microbial populations and their activities, and (iv) soil mineral N contents are required at the beginning and end of measurements. The information obtained in this way is also retrospective and applicable only to a specific soil/crop/environment combination. Nevertheless, because of its extent and, in contrast to data from other methods, availability, it may be of value in providing a guide to the ranges of mineralization likely to be encountered under a wide range of conditions.

The interpretation of N in crop removal is therefore complex. While such data integrate crop growth and soil N dynamics for any crop–climate system (Meisinger, 1982), they do not, for example, allow for any residual effects from the previous season. Further, removal into the crop is season-specific and different crops have different uptake patterns and total demand for N (Stockdale *et al.*, 1992). The below-ground plant biomass contains substantial amounts of N (Safigna, 1988) and with perennial crops there will be a reservoir of root N which does not relate to the current season's input (Huntington *et al.*, 1985). Preharvest N loss in leaf fall and/or by volatilization from leaves is also important (Huntington *et al.*, 1985) and difficult to quantify.

Nitrogen balance studies have also been used to determine mineralization either directly or by difference (Powlson *et al.*, 1986; Vinten *et al.*, 1992), and combine measurements of crop uptake, mineral N in the soil profile, N in soil organic pools, and estimates or measurements of losses. This is a very labor-intensive method and is reliant on a large range and number of measurements and analyses.

At present, information from crop N uptake studies provides the most extensive data sets for estimating mineralization, with information from fertilizer response trials for many soils, crops, and previous managements. Interpretation of these data could provide a valuable means of estimating flows of N from mineralization for a wide range of systems, but careful calibration will be required in order to estimate accurate rates of release of N.

C. LABORATORY DETERMINATION OF POTENTIAL MINERALIZATION INDICES

1. Incubation Methods

One of the most widely used approaches has been incubation of a fixed mass or volume of soil under standard conditions over a defined period (typically up to 3–4 weeks) and determination of the increases in NO_3^- and NH_4^+ concentrations. The protocols for doing this are many and varied. The objective has been to estimate the organic N pool that is available for mineralization, i.e., it is assumed that there is a pool which can be mineralized by micro-organisms in a finite time under optimal conditions. The advantage of this approach is that conditions can

be defined and held constant and some of the problems of field spatial and temporal variability can be overcome, but rates are usually higher than those measured in the field where optimum conditions rarely prevail (Adams and Attiwill, 1986). Thus, although good correlations often exist between N released during incubation and uptake in pot studies, relationships are much poorer in field studies. Many methods rely on aerobic conditions and are usually some variation of the method proposed by Stanford and Smith (1972). There are a number of modifications of this basic method. Anaerobic systems have sometimes been preferred (Keeney, 1982; Lober and Reeder, 1993) as only NH_4^+ has to be determined; because nitrification is restricted, there is no need to optimize water contents and higher temperatures can be used so that more N can be released in a given period than under aerobic conditions.

Most of the methods evaluate mineralization in soils which are much modified by mixing, sieving, drying, and/or rewetting. The possible combinations of methods/pretreatments/storage/incubation are numerous and all have impact on mineralization rates. Thus soils have been sieved (Nordmeyer and Richter, 1985), air dried (Stanford and Smith, 1972), field moist (Skjemstad *et al.*, 1988), or frozen (Beauchamp *et al.*, 1986) prior to incubation under a variety of conditions. The disruption of aggregates and disturbance of anaerobic or aerobic microsite activity can have profound impact on mineralization potential. Some methods try to reduce these problems by using intact cores. The use of structurally undisturbed cores is also difficult, in large part because the spatial variability in the field demands much replication (Macduff and White, 1985), but also because they contain excised roots which could affect mineralization kinetics (Ross *et al.*, 1985). It is clear that N mineralization is dependent on sample pretreatment and incubation conditions as well as the inherent properties of the soil and the chemical constitution of the organic materials (Nordmeyer and Richter, 1985). The conditions chosen in any one study usually reflect the particular aims of that study. Further, this approach provides a "snapshot" picture only, dependent on conditions prevailing at the time of sampling. Contents of more labile forms of organic materials change through the year, as will populations of micro-organisms. There are therefore difficulties in extrapolating results from incubation system to field scale where many factors other than those held to be important in the laboratory will be operating and/or interacting. Assumptions therefore have to be made in terms of the models used to estimate rates in the field. The use of data from laboratory incubations to predict field mineralization rates has had variable success. Where results from incubations of disturbed samples were used to predict those in the field, overestimates of 67–343% were obtained (Cabrera and Kissel, 1988). On other occasions, reasonable agreement between prediction from incubation and that measured in lysimeters has been found, but not when there were flushes of mineralization during wetting/drying sequences (Campbell *et al.*, 1988). There may also be problems where crop residues are concerned because of an unrealistic degree of mixing in incubation

systems. If the objective is to represent field conditions, then structurally undisturbed cores with much replication are more appropriate.

With all experimental approaches, models have to be developed to describe and extrapolate the information. Although used successfully in some instances, not all mineralization patterns have been described by first-order kinetics (Matus and Rodriquez, 1994; Stanford and Smith, 1972), for example zero-order kinetics were observed in fresh, undried and unamended soils (Addiscott, 1983) and in meadow soils (Simard and N'Dayegamiye, 1993). Other more complex multi-phase patterns have been described for both meadow soils (Simard and N'Dayegamiye, 1993) and plant and animal residues (Chae and Tabatabai, 1986). To determine potentially mineralizable N precisely requires that measurements are continued until the rate declines to a small constant level (Bonde and Linberg, 1988). In most instances, time dictates that mineralizable N and its associated rate factor have to be determined by fitting models to the data. It was assumed that a one-pool model (Stanford and Smith, 1972) would adequately describe the process (Section II.A), but recently two-pool models have been used with both having either first-order (Deans *et al.*, 1986) or first- and zero-order (Beauchamp *et al.*, 1986) kinetics. In fact, in the field it is likely that an exponential model with an infinite number of pools is most representative, although from a practical point of view the number of pools to be included would depend upon the objectives and desired accuracy (Sierra, 1990). Other more complex relationships have been proposed, including parabolic equations (Broadbent, 1986) and the use of the van Bertalanffy function and Weibull distributions (White and Marinakis, 1991). All of these depend on nonlinear least squares fitting of equations and produce a predicted mineralization rate as a mathematically defined quantity, and depend upon there being a homogeneous, discrete presence of organic N in the soil.

The use of incubation methods has told us much about the mechanics of the mineralization process and allowed comparison between soil types under controlled conditions. To be of further value for practical applications, this approach requires standardization, the development of a rapid method to estimate potentially mineralizable N, and the means to extrapolate the information to different cropping systems. This presents a considerable challenge but is required so that results can be used to aid advice and to add to the general understanding of the supplies of available N under field conditions.

2. Chemical Methods

a. Extractants

Chemical extractants have been used as rapid and convenient means to "quantify" N released from labile SOM. The severity of extraction varies widely from strong hydrolysis to relatively mild extraction with hot water or hot salt solutions; although the latter have shown the most promise (Stanford, 1982), acidified potassium permanganate and dichromate have been used with some success

(Wilson *et al.*, 1994). The amount of N extracted from the soil is assumed to represent, or to be related to, the N available to plants during the following growing season or to leach during winter if the extractions are undertaken in the spring or autumn, respectively. The use of such a method is therefore dependent upon correlation with crop yield, N offtake, or N leached. Some of the more recently successful methods have used KCl solutions at differing concentrations and temperatures over various times (Whitehead, 1981; Gianello and Bremner, 1986). Nitrogen released with hot 2 M KCl correlated well with N uptake by malting barley on a restricted range of Scottish soils (McTaggart and Smith, 1993). Whitehead (1981), using a similar extraction, found a good relationship with N uptake by perennial ryegrass in a pot experiment. However, the method has not been tested widely and, while it might provide an index of the basal rate of mineralization (Stockdale and Rees, 1994), it has not predicted the changes resulting from recent additions of plant or animal residues (McTaggart and Smith, 1993). None of the proposed methods or derived indexes has been adequately tested under a broad range of field conditions or put to general use.

b. Labile N

Another recent method has been the use of electro-ultrafiltration in which anions and cations in soil suspensions are separated in an electrical field. The solutions recovered from this treatment contain not only NO_3^- and NH_4^+ but also soluble organic N. They are therefore assumed to contain readily mineralized organic N, i.e., all forms of N that are likely to be available for plant uptake or transformation processes over the shorter term. The method has been used in some European studies to provide better prediction for fertilizer requirements (Mengel, 1991) and has in some instances provided good correlation with N uptake and N mineralized from soils in pot experiments with rape (Appel and Mengel, 1990). Other relationships have been developed for the supply of N to barley in field trials and EUF extracts (and also with UV absorption at 205 nm) (Linden *et al.*, 1993). In other studies, the technique has provided poor correlation with N uptake by maize from unfertilized plots (Saint Fort *et al.*, 1990) and also where fresh residues have been incorporated (Appel and Mengel, 1993). While there may be a useful role for this technique to improve short-term prediction of N supplies and early season utilization of N, it is unlikely to be relevant to a whole season or to have general applicability across a wide range of soils/sites.

c. Soil Organic Matter/Total N

It has been suggested (Hadas *et al.*, 1986) that a good general approach is to use total soil organic matter N content as an index of the potential to supply N. On theoretical grounds this would seem unlikely given the nonuniform nature of the substrates and the wide range of soil and environmental conditions to which they are subjected. There may be some value in using this approach to examine extremes, for example highly organic versus mineral soils, but the need to do this

is restricted. There have been only a few recent attempts to use this approach and often there has been little or no relationship between total soil N with either uptake or estimated mineralization rates (Appel and Mengel, 1990). Freytag *et al.* (1989) showed that although there were no simple relationships, by grouping the soils by site and/or by some soil characteristics, a degree of success in estimating mineralization could be achieved. To provide sufficient confidence that the method could be used for a wide range of cropping/soils/environment conditions would require substantial input.

d. Extractions for Residues and Organic Wastes

There are many reports of methods to assess the likely immediate impact of manures and plant residues on mineral N release. Parker and Sommers (1983) found that the use of chemical indices oversimplified the complex process of N release from manures. Various methods have been used, e.g., acidified and alkaline permanganate in aerobic incubations (Castellanos and Pratt, 1981; Parker and Sommers, 1983), autoclaving (Parker and Sommers, 1983; Douglas and Magdoff, 1991), 6 M HCl (Serna and Pomares, 1991), pepsin digestion (Castellanos and Pratt, 1981), Walkley-Black digestion (Douglas and Magdoff, 1991), and CO₂ respiration during incubation (Castellanos and Pratt, 1981), with varying degrees of success. C:N ratio, %N, and phenolic content of residues have been identified as characteristics which influence mineralization, but there is little consistency in their relative importance (Section III.A). Whereas in some studies (Frankenberger and Abdelmagid, 1985) there was a strong correlation between residue N content and mineralized N, in others (Silvapan *et al.*, 1985) soluble N content was found to be more important. On their own, lignin or phenolic contents have not always good been predictors of N mineralized but have helped when integrated with some means of assessing N content (Fox *et al.*, 1990; Silvapan *et al.*, 1985). While C:N values are often available, these give variable results when used as a sole factor to explain mineralization rates.

While much is to be learned from the use of many of the methods noted above, the general applicability of the information that each achieves is likely to be restricted. Where residues are involved there is a need to define the various interactions between their characteristics and soil/environment conditions. There is no suitable database for either SOM or residue characteristics to permit this definition.

D. FIELD MEASUREMENTS

1. Changes in Mineral N

Various studies have attempted to deduce some measure of mineralization by following the changes in soil contents and/or losses of NO₃⁻ and NH₄⁺ over time.

Although useful, this is a simplistic approach and is not likely to provide a realistic assessment because of the internal recycling, transformation, and external inputs and losses that may occur. Mineralization rates calculated from observations of change in size of the mineral N pool should measure all of the processes or aim to exclude them. It is also worth noting that the size of the pool does not necessarily reflect its importance. Thus, although the NO_3^- pool is often very small at any one time, it can turn over in 1–2 days and maintain available supplies (Davidson *et al.*, 1990). Good correlations between mineral N measurements made in the topsoil at sowing of spring barley and N uptake of the crop (McTaggart and Smith, 1993) have been shown. It was suggested that, by the time the soil samples were taken, some mineralization had already occurred and this gave a good indication of the site differences in mineralization which would have persisted throughout the season. A major problem in using changes of mineral N to measure mineralization rates is the degree of spatial variability that exists in soils and the difficulty of making comparisons over time with spatially separated samples. Even under uniform management mineral N is spatially highly variable; if this is increased by residue addition or excretal returns under grazing, the problem is confounded. It has been estimated that, depending on the area involved, between 24 and 40 individual samples per field are required for analysis (Goovaerts and Chiang, 1993; Stockdale *et al.*, 1994). A further complication is that mineral N contents are often log-normally distributed (Macduff and White, 1984; Stockdale *et al.*, 1994). Grassland soils provide a further complication because (i) they support a perennial crop with potential for removal of mineral N throughout the year whenever environmental conditions are appropriate and (ii) they may accumulate large amounts of mineral N through recycling in excreta (Jarvis, 1992) either at grazing or after application of farm wastes.

2. Field Incubations

A range of field incubation techniques has been proposed with varying degrees of sophistication. There has been a recent resurgence of interest in these techniques because of the current need to understand all inputs and internal cycles of N under practical conditions. Techniques have included using a soil sample sealed into a polythene bag and repositioned back into the soil under ambient conditions, i.e., at the same temperature, with minimal change to soil moisture status and no risk of interaction with plants (Gordon *et al.*, 1987). In other measurements intact cores have been used (Alves *et al.*, 1993) to reduce the changes due to disturbance either in polythene bags (Nadelhoffer *et al.*, 1985) or in capped tubes (Raison *et al.*, 1987). The latter system has been refined to include exchange resins to trap any NO_3^- leaving and, on occasion, entering the columns so that changes in soil NO_3^- and NH_4^+ could be determined more accurately (DiStefano and Gholz, 1986).

Another recent method has been developed to help reduce the errors incurred because denitrification removed NO_3^- . Intact cores were incubated in sealed containers in the field at ambient soil conditions with acetylene added to inhibit nitrification and thus reduce the opportunity for denitrification (Hatch *et al.*, 1990). This has been used on a regular, routine basis (Hatch *et al.*, 1991; Gill *et al.*, 1995) to provide estimates for net mineralization in grassland soils. A further degree of refinement was introduced by Blanton (1991) who incubated intact cores sheathed in plastic to minimize effects on the core surface through changes in gaseous composition and in an atmosphere of 20% O_2 in helium after purging N_2 from the soil atmosphere so that full account could be taken of denitrification effects.

3. Applicability of Field Methods

Field methods have the advantage that they allow measurement under ambient conditions over long periods, are relatively simple to use, make some attempt at reducing soil variability, and take steps to reduce or stop losses. On the other hand, there is a requirement for frequent measurements and a need for some assurance that, where it is used, acetylene is effective and that atmospheric conditions within the vessel do not deviate too far from those present originally. Furthermore, little attempt is made to control water status which is only representative of what is present at the start of a measurement period. Ideally, large numbers of replicates are needed (Raison *et al.*, 1987). Few cross comparisons between methods have been made, but Redman *et al.* (1989) compared net mineralization rates determined with field core incubations with those derived from a budget study. They found that while there were no differences during winter, there were significantly higher rates from the incubation methods between April and July. Undisturbed cores in plastic bags compared with exposed cores with ion-exchange resins gave similar estimates (Zou *et al.*, 1992). All methods, even where cores are left intact, are affected by sampling and incubation procedures which will have effects on net N release, including that derived from damaged roots. It seems unlikely that any one method could ever fulfill all needs and that a range of approaches will have to be utilized.

4. Use of ^{15}N -Labeled Fertilizer

By labeling inorganic N fertilizer with ^{15}N it is possible to distinguish between N derived from fertilizer and that derived from other sources which, in most cases, will be through mineralization. Thus, in principle, a measurement of unlabeled N in the crop at the time of harvest plus the net change in unlabeled inorganic N in the soil should provide a measure of net mineralization.

This is an extension of the "zero N plot" method already discussed and it is

arguable whether the data derived from a ^{15}N experiment in this way are superior to those from the zero N plot method. The values obtained from the two methods very often differ, with the uptake of unlabeled N in ^{15}N experiments often exceeding the total N uptake by a crop given no N fertilizer under the same environmental conditions (e.g., Powlson *et al.*, 1986, 1992). This has been described as a priming effect, with the implication that the addition of fertilizer N stimulated the mineralization of native organic N. However, in most cases, the difference is likely to arise through pool substitution, in which labeled inorganic N substitutes for a part of the unlabeled inorganic N that would otherwise have been immobilized or lost through denitrification (Jenkinson *et al.*, 1985). Although this results in a larger amount of unlabeled N being available for crop uptake, it is an artifact of the experimental procedure as discussed in detail by Hart *et al.* (1986).

If ^{15}N labeling is used to assess soil N supply through mineralization, the data must be interpreted with great caution. Data showing crop uptake of unlabeled N are usually obtained as a by-product of ^{15}N experiments with other objectives, e.g., following the fate of fertilizer N in a specific system (e.g., Hart *et al.*, 1986), to construct N balances in order to assess overall loss of fertilizer N (e.g., Powlson *et al.*, 1986, 1992), or to obtain estimates of gross mineralization as described below (Section IV.E).

E. MEASUREMENT OF GROSS MINERALIZATION

The importance of understanding gross mineralization has already been indicated (Section I.B). Technology has provided the ability to distinguish between stable isotopes of elements such as N with relative ease and has opened up opportunities to examine soil N processes with considerable precision. Monitoring changes in enriched sources of N added to soils allows a good deal of sensitivity, precision, and positive identification of labeled forms of N as they enter, are transformed in, or leave the system under study. There are limitations due to the biological internal recycling occurring in soil systems, and interpretation of ^{15}N data thus needs considerable care. Two general approaches have been used. The first, noted above (Section IV.D) involves N balance studies with enriched N added with fertilizer, for example, and with budgets for all the appropriate inputs and outputs. The degree of dilution of the enrichment within the system provides some estimate of input from other sources, i.e., through mineralization (Nason and Myrold, 1991). Second, ^{15}N labeling has been used to determine specific rates of transformation (Myrold and Tiedje, 1986; Barraclough and Smith, 1987). Thus the rate that added enriched $^{15}\text{NH}_4^+$ is diluted as unlabeled organic N is mineralized and is added to the soil NH_4^+ pool provides, over the short term, a very accurate measure of gross mineralization rate.

By labeling with $^{15}\text{NH}_4^+$ or $^{15}\text{NO}_3^-$ and examining soils after a relatively short period, rates of mineralization as indicated by the decline in ^{15}N enrichment of NH_4^+ , of nitrification by the decline in ^{15}N enrichment of NO_3^- , and of consumption/removal or immobilization of NH_4^+ and NO_3^- by the disappearance of ^{15}N label have all been determined on a limited range of soils (Barracough, 1991; Barracough and Smith, 1987; Myrold and Tiedje, 1986). However, there are a number of assumptions implicit in the use of these methods, i.e., (i) the processes occur at constant rates, or vary in a known way, between sampling times, (ii) there is no rapid internal cycling, and (iii) the processes exploit ^{15}N and ^{14}N in all available pools in proportion to their relative amounts.

Development of stable isotope methodology for N, especially if linked to similar studies with carbon, can contribute much to our understanding of the complexity of soil mineralization processes. The problems of execution and interpretation are numerous. As well as the assumptions above, it is necessary to assume that ^{14}N and ^{15}N are not discriminated by soil processes and that any added ^{15}N equilibrates rapidly with the pool to which it has been added. It is known that some microbial processes do discriminate between ^{14}N and ^{15}N (Heaton, 1986). The heterogeneity of soils, especially the spatial variability of NH_4^+ and NO_3^- , presents a serious problem in obtaining, first, a uniform distribution (Barracough, 1991) and, second, the development of *in situ* methods of isotope dilution, since rate estimates are calculated by differences which amplify errors (Myrold and Tiedje, 1986). $^{15}\text{NH}_4^+$ injected or mixed with soils is assumed to be uniformly distributed but will inevitably enter macropores and be spatially separated from much of the background soil solution, exchangeable and fixed NH_4^+ . Added $^{15}\text{NH}_4^+$ and native $^{14}\text{NH}_4^+$ may then be subjected to different consumption rates leading to errors in calculation of gross mineralization.

The use of pool dilution techniques, therefore, is not always straightforward. Continued, sensible use and reassessment of the procedures involved should, however, lead to an increased understanding of the supplies of mineral N from native soil sources. Increased definition of gross mineralization rates of recently returned organic materials as compared with "old" native SOM is an essential prerequisite to manipulating existing systems or devising novel management systems to control and utilize N from mineralization. Most information can be obtained where pool dilution experiments are coupled with direct tracer methods, where the fate of added ^{15}N is determined at the end of the experiment (Powlson and Barracough, 1993).

F. THE ROLE OF MODELS IN PREDICTING MINERALIZATION

Most models of N flows in cropping systems incorporate some consideration of mineralization but generally lack a mechanistic basis. Model design depends

on intended use, how much detail is used to describe the subsystems, and the way in which they are influenced by environmental factors. Choice of an appropriate time step is also important, and controls the amount of data needed to run the model and the nature of the output. Specific processes are often excluded because there are insufficient reliable data available either to derive or to validate the relationships; the release of N from manures is a good example of this. Models intended for widespread use as management tools may be restricted in their design due to limited available data inputs or they may have to incorporate default data sets.

1. Practical Models

One of the main problems is to decide at what level of detail to model the soil system: this will be dependent on the intended use of the model. Management models to be used in policy development or by farmers and advisors have to use inputs which are relatively easily available and which are likely to be on a field or hectare basis. The intricacies of the soil N cycle at the microsite scale then seem to be relatively unimportant, but most microsite processes, e.g., denitrification, which is dependent on local O_2 and soluble C and NO_3^- concentrations, are important at field scales (Section III.D). It is important to be able to estimate the effects of field scale management either by studying the literature or by running a model which operates at an appropriate scale to derive a simple relationship. The amount of detail included in models designed to be used as management tools varies enormously. However, the prediction of the amount of N released by mineralization by more complicated models has not given better results than simpler ones (de Willigen, 1991).

Nitrogen management models are used to give immediate fertilizer recommendations, to determine the likely impact of management changes on farm profitability or environmental impact, and to allow development of good agricultural practice to minimize losses. These models should have mineralization as an important subsystem to determine the release of N from crop residues, manures, and SOM. The detail at which mineralization, and the other subsystems with which it interacts, is described will vary between models.

2. Research Models

Research models try to integrate new understanding about soil processes and their interactions. These models may be very different from each other and may use entirely new approaches but have an important role in increasing our understanding of mineralization and its relationship with other components of the soil N cycle. Such models enable both synthesis of new information from experimental data and confirmation or rejection of hypotheses about how the system oper-

ates, and highlight gaps in knowledge. Research models seem certain to become larger and larger as the importance of more and more subsystems is recognized. The development of a mathematical representation of soil porosity is crucial to allow the modeling of the interrelationships between soil structure, soil biota, and nutrient cycling (Juma, 1993). Changes in the output of models caused by the introduction of new subsystems may be very small: often the effect is to alter the pathways and partitioning of N within the SOM pools or to add new pools. However, future changes in the perception of microsite activities may influence the way the whole system responds to environmental perturbations and so should not be disregarded when a management model is being designed or reevaluated. If management models are to be accurate and scientifically credible they must take note of new concepts, derived from both modeling and experimental approaches, at the research level. New, and often complicating, factors can then be included explicitly or implicitly in a simplified way or excluded on the basis of a rational judgement. A different recent approach has been to model N mineralization from the trophic interactions among the groups of organisms constituting the soil food web which produces results close to *in situ* measurements (Ruiter *et al.*, 1993).

V. THE IMPACT OF MINERALIZATION

In this review we have shown that the effects of mineralization are considerable and widespread. Current issues related to excesses of N in agricultural management being transferred to other systems demand a knowledge of, and an ability to predict, the flows of N from all sources. There is little doubt that supplies from native sources are substantial, but direct measurements of mineralization for most conditions are few and far between. It is essential to the development of useful models that the current database is extended to a wider range of soil types and agricultural managements. This could then be coupled with the existing information for soil and climate and used to provide more field-specific advice.

The rates of transfer in grassland can be particularly large. There have been only a few studies which have measured either gross or net mineralization directly under field conditions. Geens *et al.* (1991) measured gross mineralization and immobilization rates under long term grass on a clay loam soil. In April, gross mineralization rates were 2–3 kg N ha⁻¹ day⁻¹ and gross immobilization rates were 1–5 kg N ha⁻¹ day⁻¹, while in June mineralization rates had increased to 1–5 kg N ha⁻¹ day⁻¹ and immobilization rates were 1–3.5 kg N ha⁻¹ day⁻¹. Recently, three studies have determined net mineralization in grazed pastures through extended periods. Average daily rates of net release ranged from 0.2 to

2.3 kg N ha⁻¹, depending on background sward management (Table IV). As well as the direct effects of the very large rates of N released and available for plant uptake or loss, it is of interest to note that in the study of Gill *et al.* (1995) an average of 85 kg N ha⁻¹ was released during the period November to February, i.e., when opportunities to be leached were at their greatest in the UK. Substantial winter mineralization rates have also been observed in grassland soils in the Netherlands (Olf *et al.*, 1994).

Disturbance of grassland can have substantial impact. Older swards generally mineralize more N after disturbance than short-term leys, because of greater N accumulation, but there is always a declining release of N which continues for decades. Young (1986) calculated that N mineralization increased from about

Table IV
Net Mineralization Rates in Grassland Soils

Site	Soil type	Fertilizer N management (kg ha ⁻¹)		Measurement period	N mineralization	
		Past	Present		kg ha ⁻¹	kg ha ⁻¹ day ⁻¹
SE England ^a	Sandy loam	0	0	April–Oct	415	2.3
		420	0	April–Oct	321	1.8
		420	420	April–Oct	310	1.7
SW England ^b	Silty clay loam, drained	400	400	12 months	314	0.9
	Silty clay loam, drained	0	0	12 months	162	0.4
	Silty clay loam, undrained	400	400	12 months	173	0.5
	Silty clay loam, undrained	0	0	12 months	65	0.2
SW England ^c	Silty clay loam, drained	200	200	12 months	376	1.0
	Silty clay loam, undrained	200	200	12 months	317	0.9
	Silty clay loam, undrained	200	0	12 months	292	0.8
	Silty clay loam, undrained	0	0	12 months	135	0.4
	Silty clay loam, undrained	0	200	12 months	270	0.7

^a Hatch *et al.* (1991).

^b Blanter (1991).

^c Gill *et al.* (1995).

100 kg N ha⁻¹ after ploughing a 1-year ley to 280–380 kg N ha⁻¹ for leys older than 4 years. Whitehead *et al.* (1990) measured N accumulation in stubble, leaf litter, roots, and soil macro-organic matter, which was 536 kg N ha⁻¹ for an 8-year ryegrass sward and 602 kg N ha⁻¹ for a 15-year-old sward. They also demonstrated that management, in particular N inputs and grazing, influenced N accumulation and, hence, mineralization after ploughing. Williams and Clement (1965) had previously shown this with incubation studies. Shepherd (1993) found that the use of soil mineral N sampling to predict N response of wheat after ploughing grass swards was of no value because subsequent immobilization/mineralization processes were too variable. Manure application appeared to increase N mineralization potential on ploughing out grass (Withers, 1988). Increased stocking rates, stimulated by increased fertilizer use, had a large effect on potential N mineralization (Lloyd, 1992). N mineralization from fields cut for silage has been reported as being negligible (e.g., Clement, 1975; Withers, 1988) but depended very much on the previous N balance of the sward. Large amounts of manure or slurry are often applied during the winter before ploughing out in the UK and this will further affect mineralization. Clover-based swards may be a special case (Section II.E) and have a specific effect in increasing mineralization on ploughing out grassland. This may become more important if increased symbiotic N fixation within farming systems results from the search for increased sustainability. In general, although the amount and timing of N release on ploughing grassland are known qualitatively, the relationships need further quantification.

In arable systems, uptake of N from soil and plant residue sources forms a significant part of total crop N uptake. In experiments using ¹⁵N fertilizers to measure fertilizer use efficiency, unlabeled N uptake assumed to be that from the soil has been shown to be significantly different between soils, even where fertilizer N uptakes are similar (McTaggart and Smith, 1993). This may be due to the effect of previous crop residues: increased uptake of soil N occurred where the residues of the brussel sprout crop were involved (Table V). However, at other sites where cereals were the previous crop, significant differences in soil N uptake were seen. In most, but not all, cases, uptake of soil N increased as the SOM content increased. Understanding rates of supply from N catch crops is an important issue in relation to potential leaching (Thorup-Kristensen, 1993) but little is known about long-term effects of this.

Understanding N mineralization from manures is particularly important to provide better information about short-term N supply, but there are important longer term implications because of contributions to fertilizer value in the following years [discussed elsewhere by Shepherd *et al.* (1996)] and the need to avoid increased leaching risk with time. Thus the mineralizable N pool can be increased over the long term by frequent additions. For example, N uptake by plots receiving no mineral fertilizer has been shown to vary annually because of

Table V
Nitrogen Uptake (kg ha^{-1}) of Spring Barley Grown on Six Sites
in Southern Scotland in 1990 Divided into That Derived from
Fertilizer and That from the Soil (McTaggart and Smith, 1993)

Previous crop	Soil texture	OM %	Nitrogen uptake (kg N ha^{-1})	
			From fertilizer	From soil
Cereals	Sandy loam	2.4	63.9	47.2
Brussels sprouts	Loamy sand	2.8	73.3	122.8
Cereals	Sandy clay loam	3.3	51.0	49.1
Cereals	Sandy loam	4.7	70.0	71.3
Cereals	Clay loam	5.1	46.4	78.6
Cereals	Sandy loam	5.7	58.5	41.1

weather differences, but the variation was greater where the amount of mineralizable substrate was larger, in plots treated annually with farmyard manure. Similarly, the application of fertilizer N for many years increased mineralizable soil N in many long- and medium-term experiments (Glendining and Powlson, 1995) by increasing the amount of crop residues returned to the soil. The increases were generally substantial, i.e., 20% or more compared with the control plots and tended to be greater than any increases in total soil N.

Residues returned and manures added to the soil are particularly important by acting as a source of generally readily mineralized N for a following crop, and also as a major source of N lost from the system. Optimization of when and how a residue or manure is returned to the soil is important for efficient use of the resource. Although there is much qualitative information about these processes, this will require a transformation of this knowledge and of the controlling factors to quantitative principles which can be used to improve the construction of predictive models and, thus, advice.

The current, major practical objective of obtaining information on mineralization is to provide the basis on which to achieve increased efficiency/reduced environmental impact in agricultural systems. Inevitably, attempts to do this have used methods which integrate the effects of many interactive processes and controlling factors by measuring, for example, changes in mineral N. The detail with which the N cycle processes are viewed decrease as the hierarchy of scale increases from microsite through soil profile, field to catchment, regional and global levels, with the primary difference between scales being the complicating effect of spatial variability (Fig.3). Spatial variability of the processes and their products occurs both laterally and with depth and makes it difficult to calculate an appropriate field "average" from data collected at individual points (Biggar,

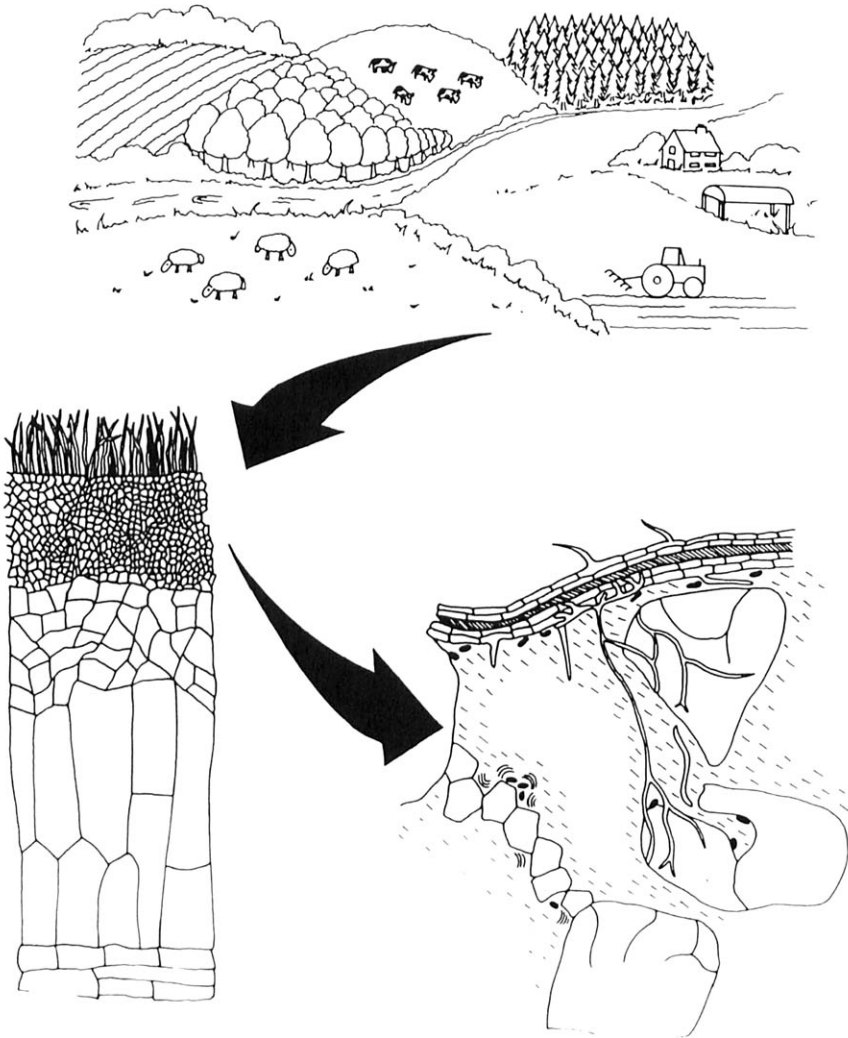


Figure 3 Mineralization can be considered at a range of scales: shown here are catchment, soil profile, and microsite. The detail at which the processes are seen increases as the hierarchy descends from landscape to farm, to soil profile toward the microsite scale.

1978). This problem arises whenever there is a need to transfer information between any two levels in the hierarchy. Relationships between different levels are difficult, but important, to establish, so that the results of research can be put into practical applications. From the outset, the success of transposing information from one scale to another will depend upon (i) better process definition and

(ii) improved methods of making estimates of their effects. Predictive or advisory tools that aim to involve mineralization will be of more functional use if based on an improved understanding of the distributions of biological activities in relation to microsites and soil mineral N pools is obtained (Ladd *et al.*, 1993).

VI. CONCLUSIONS AND FUTURE PROGRESS

The overall effect of all the processes involved in the net release of available N has implications for all ecosystems, whether managed or natural, and at a range of scales from microsite to global. It is clear from this review that while much information exists about the process generally known as mineralization, we are a long way from being able to either maximize its potential or reduce its impact. The very basis of being able to do this will be an increased understanding at the process level and an improved ability to measure the results of those processes. The following points summarize the current status of the problems involved and indicate the way in which we consider that progress can be made.

- Better quantification of the role of mineralization would (a) help minimize losses of N to the environment, (b) enable better fertilizer advice, and (c) improve management of N within the crop–soil system. While there is some basis for this, detailed information is restricted to a limited number of situations. At global, national, regional, and farm scales, changes in policies and agricultural management for which there is no current advice will demand greater knowledge and predictive capacity for mineralization.

- Increased efficiency of N use will demand that a full account is taken of the contribution of mineralization and immobilization to N budgets at field, farm and catchment scales. This will require not only a synthesis of current available knowledge, but in many instances the development of new databases for both short- and long-term effects. There will be an urgent need for more data from new managements such as set-aside, for new crops and ley-arable farming systems as practices respond to political, environmental, and socioeconomic pressures.

- There is a need to (a) examine the impact of management practices, e.g., cultivation and residue distribution, on soil properties at the microsite level, (b) determine the influence of these changes on biological activity, and (c) determine the extent to which variability at the microsite level influences the flows of N at a field scale. All of these will depend upon an appreciation of the new concepts of the way in which soil architecture interacts with spatial distributions of biological activities and specific N pools and a knowledge of how diffusional constraints affect the processes in widely heterogeneous microsites. The new techniques

available for measuring gross rates of N mineralization will provide a foundation on which to base the principles for more applied considerations.

- Mineralization controls much of the NO_3^- losses by leaching. However, the contribution of mineralization to other losses, e.g., denitrification, need to be quantified (Seitzinger, 1994). Other specific areas requiring attention include the role of nitrification as a limitation to leaching and denitrification, and the contribution of mineralization of SOM and leached organic N at depth to NO_3^- leaching into subsurface aquifers.

- It should be possible to reexamine crop N uptake in past N response trials to provide immediate information on N supplies from native sources for the most common crops. This could then be modified by any future improvements in direct methodology or approaches to mineralization estimates.

- Detailed experiments are required to measure gross and net mineralization for representative sites across a range of crops, climates, and soils. A comprehensive understanding of N flows would enable the principles developed to be applied with appropriate models to a full range of crops/soils.

- The identification of pools within both SOM and added residues which can be readily separated from soil and analyzed and which have biological relevance will be important. The quantification of C and N in these pools and their interactions will enable better understanding of mineralization/immobilization and more precise definition of resource quality in relation to the amounts and timing of N released. It will also provide more appropriate definition of pools within models and greater possibilities for testing model simulations.

- At the same time, the development of field methods for measuring net mineralization is important and requires the selection of the most appropriate technique to produce robust information. In the first instance, there will be a need to reexamine whether a simple test (e.g., soil extraction or incubation) is a feasible way to predict N mineralization. Even if this approach is not universally applicable it may still be of value in a significant number of situations. Data will also be required which define the factors controlling soil N supply including the effect of season and long-term management history.

- Finally, the long-term nature of the process and its effects should be emphasized. The full impact on mineralization of, for example, a change in agricultural practice may be apparent only after some years and the longer-term results may be very different from those obtained immediately.

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