

# Nitrogen Mineralization from Organic Residues: Research Opportunities

M. L. Cabrera,\* D. E. Kissel, and M. F. Vigil

## ABSTRACT

Research on nitrogen (N) mineralization from organic residues is important to understand N cycling in soils. Here we review research on factors controlling net N mineralization as well as research on laboratory and field modeling efforts, with the objective of highlighting areas with opportunities for additional research. Among the factors controlling net N mineralization are organic composition of the residue, soil temperature and water content, drying and rewetting events, and soil characteristics. Because C to N ratio of the residue cannot explain all the variability observed in N mineralization among residues, considerable effort has been dedicated to the identification of specific compounds that play critical roles in N mineralization. Spectroscopic techniques are promising tools to further identify these compounds. Many studies have evaluated the effect of temperature and soil water content on N mineralization, but most have concentrated on mineralization from soil organic matter, not from organic residues. Additional work should be conducted with different organic residues, paying particular attention to the interaction between soil temperature and water content. One- and two-pool exponential models have been used to model N mineralization under laboratory conditions, but some drawbacks make it difficult to identify definite pools of mineralizable N. Fixing rate constants has been used as a way to eliminate some of these drawbacks when modeling N mineralization from soil organic matter, and may be useful for modeling N mineralization from organic residues. Additional work with more complex simulation models is needed to simulate both gross N mineralization and immobilization to better estimate net N mineralized from organic residues.

**N**ITROGEN MINERALIZATION and immobilization are important processes in the N cycle. Nitrogen mineralization is the conversion of organic N into ammonium N, whereas N immobilization is the conversion of inorganic N into organic N (Alexander, 1977). Both processes occur simultaneously in soil, with the relative magnitudes determining whether the overall effect is net N mineralization or net N immobilization.

Understanding the role of these processes when organic residues are applied to land is important to understand N cycling in soils and to develop models that estimate the amount of inorganic N that may become available to crops (Quemada and Cabrera, 1997a). Such capability is necessary to achieve long-term sustainability in the use of organic residues. In this work, we review research on factors controlling net N mineralization as well as research on laboratory and modeling efforts, with the objective of highlighting areas with opportunities for additional research.

M.L. Cabrera and D.E. Kissel, Crop and Soil Sciences Department, University of Georgia, Athens, GA 30602-7272. M.F. Vigil, USDA-ARS, Akron, CO 80720-1029. Received 31 Mar. 2004. \*Corresponding author (mcabrera@uga.edu).

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677 S. Segoe Rd., Madison, WI 53711 USA

## FACTORS CONTROLLING NET NITROGEN MINERALIZATION

Among the factors controlling net N mineralization are organic composition of the residue (Whitmore, 1996), soil temperature and water content (Kätterer et al., 1998), drying and rewetting events (Kruse et al., 2004), and soil characteristics (Schjønning et al., 1999; Gordillo and Cabrera, 1997b).

## Composition of Organic Residues

Organic residues added to the soil surface or incorporated into the soil undergo decomposition by the microbial biomass present in soil and/or residues. Part of the carbon in the decomposing residues is evolved as CO<sub>2</sub> and part is assimilated by the microbial biomass involved in the decomposition process (Alexander, 1977; Gilmour et al., 2003). For the assimilation of C to occur, N also has to be assimilated in an amount determined by the C to N ratio of the microbial biomass. If the amount of N present in the decomposing organic residue is larger than that required by the microbial biomass, there will be net N mineralization with release of inorganic N. If the amount of N in the residue is equal to the amount required there will be no net N mineralization. If, on the other hand, the amount of N present in the residue is smaller than that required by the microbial biomass, additional inorganic N will need to be immobilized from the soil to complete the decomposition process (Corbeels et al., 1999).

The above discussion suggests that the amounts of C and N in residues and in decomposing microbial biomass are important factors controlling the occurrence of net N mineralization or net N immobilization. Research summarized by Whitmore (1996) clearly shows that the C to N ratio of residues is related to the amount of N released and that the break-even point between net N mineralization and N immobilization can be found between C to N ratios of 20 and 40. Similar results were reported by Van Kessel et al. (2000), Seneviratne (2000), and Qian and Schoenau (2002). In some cases, however, the break-even point has been found to be near 15 (Gilmour, 1998). The existence of a range instead of a single value for the break-even point is probably related to variation in the C to N ratio of the decomposing microbial biomass as well as the existence of organic components with different susceptibility to decomposition. Organic residues with similar C to N ratios may mineralize different amounts of N because of differences in composition that are not reflected by the C to N ratio. Because C to N ratio by itself cannot explain all differences in N mineralization, much effort has been spent on characterizing the different compounds or groups of compounds present in organic residues. Some studies have identified groups such as polyphenols, proteins, soluble carbohydrates, and hemicellu-

lose-like, cellulose-like, and lignin-like compounds, and have related them to residue decomposition (Thuriès et al., 2002) and N mineralization (Palm and Sanchez, 1991; Vigil and Kissel, 1991; Lerch et al., 1992; Constantinides and Fownes, 1994; Seneviratne, 2000; Rowell et al., 2001). Other studies have characterized organic residues using different spectroscopic techniques. For example, Rowell et al. (2001) related N mineralization from four biosolids, wheat straw, paper fines, and needle litter to the initial chemistry, which was characterized by  $^{13}\text{C}$  nuclear magnetic resonance (NMR) spectroscopy. They found that net N mineralized was more strongly related to alkyl-C content ( $r = 0.86$ ) than C to N ratio ( $r = -0.75$ ). Similarly, Qafoku et al. (2001) showed that characterization of poultry litter by near infrared reflectance spectroscopy may be useful to estimate potentially mineralizable N ( $r = 0.90$ ). A few studies have found good relationships between potentially mineralizable N and a water-soluble fraction in the organic residue. For example, Qafoku et al. (2001) reported that potentially mineralizable N was strongly correlated with water-soluble organic N ( $r = 0.93$ ) and poorly correlated with C to N ratio ( $r = -0.26$ ). Similarly, De Neve and Hofman (1996), working with vegetable crop residues, reported that mineralizable N was better correlated with the water-soluble fraction in the residues ( $r = 0.86$ ) than with C to N ratio ( $r = -0.76$ ). Finally, a few studies have identified specific compounds that are strongly related to N mineralization. For example, in a study with 15 poultry litter samples, Gordillo and Cabrera (1997a) found that mineralizable N was strongly correlated with uric acid content ( $r = 0.90$ ). It is clear from these results that while C to N ratio is an important variable, other composition variables also play important roles in the N mineralization process. Research in this area should continue to identify specific N compounds or groups of N compounds that affect N mineralization in different types of organic residues. Spectroscopic techniques seem to offer good research tools for that purpose.

Other unique properties of some organic residues, such as pH, salinity, and heavy metal concentration, may also affect microbial activity and N mineralization. For example, the addition of heavy metals to sewage sludge in some cases has increased N mineralization (Hassen et al., 1998; Khan and Scullion, 2002) and in some other cases has decreased N mineralization (Dar and Mishra, 1994). Additional research is needed to understand the mechanisms involved.

### Soil Temperature and Water Content

Many studies have evaluated the effect of soil temperature and water content on N mineralization (Stanford and Epstein, 1974; Ellert and Bettany, 1992; Gonçalves and Carlyle, 1994; Sierra, 1997), but most have concentrated on mineralization of N from soil organic matter, not from organic residues (Griffin et al., 2002). Also, many studies have evaluated the effect of temperature at optimum water content (Ellert and Bettany, 1992) or the effect of water content at optimum temperature (Stan-

ford and Epstein, 1974; Myers et al., 1982), implicitly assuming no interaction between these variables. Many studies have shown, however, that there is an important interaction between temperature and water content with regard to N mineralization (Gonçalves and Carlyle, 1994; Sierra, 1997; Knoepp and Swank, 2002). Quemada and Cabrera (1997b) found a strong interaction between temperature and water content in the mineralization of N from surface-applied crimson clover (*Trifolium incarnatum* L.) residues. They also found that the effects of temperature and water content on N mineralization from soil organic matter were different from the effects of temperature and water content on N mineralization for surface-applied residues. Additional work is needed with other organic residues to further understand the interaction between temperature and water content in incorporated and surface-applied residues.

### Drying and Rewetting Events

Large changes in soil water content caused by drying and rewetting events constitute another environmental factor that may have an important effect on N mineralization. Drying and rewetting effects on N mineralization from soil organic matter have been extensively studied (Birch, 1958; Agarwal et al., 1971; Cabrera, 1993; van Gestel et al., 1996; Appel, 1997), but limited data are available on the effect that drying and rewetting may have on N mineralization from organic residues (Clein and Schimel, 1993; Pulleman and Tietema, 1999; Magid et al., 1999; Kruse et al., 2004). In a recent study, Kruse et al. (2004) found that cotton (*Gossypium hirsutum* L.) leaves decomposing in continuously moist soils for 185 d led to mineralization of 30% of the applied N, whereas cotton leaves decomposing in soil subjected to a 14-d drying-rewetting cycle for 185 d led to minimal net N mineralization or net N immobilization. The authors hypothesized that this effect resulted from the effect of drying and rewetting on macrofauna populations that prey on bacterial populations. Clein and Schimel (1993) reported lower decomposition in dried and rewetted birch litter than in continuously moist litter. They hypothesized that this reduction was caused by the drought sensitivity of a key microbial population in the litter. Similarly, Magid et al. (1999) found that drying and rewetting decreased the decomposition of added perennial ryegrass (*Lolium perenne* L.) shoot material.

Drying and rewetting events are likely to have a more marked effect on surface than on incorporated residues, and are probably responsible for the slower decomposition and N mineralization rates observed in surface residues (Schomberg et al., 1994). Additional work with different organic residues is needed to further understand the mechanisms involved in drying and rewetting events in both surface and incorporated residues. This work should include evaluation of the microbial population and microfauna involved in the decomposition process.

### Soil Characteristics

Some laboratory studies on N mineralization from organic residues have shown differences in the amounts

of N released from the same residue in different soils (Whitmore and Groot, 1997; Gordillo and Cabrera, 1997b; Thomsen and Olesen, 2000). Differences in N release may be attributed to adsorption of organic N by clays (Van Veen et al., 1985), increased aeration in sandier soils (Thomsen et al., 1999), different C to N ratios of microorganisms and microfauna (Hassink, 1994; Hassink et al., 1994), and different populations of protozoa and nematodes present (Kuikman et al., 1991; Griffiths et al., 1998). In a study with poultry litter, Gordillo and Cabrera (1997b) found that the amount of N mineralized in nine different soils was related to the ratio of sand content to (water content at field capacity). The ratio of sand content to (water content at field capacity) is related to soil aeration and to water-filled pore space suitable for the microbial and microfauna populations present in these soils.

Other soil characteristics, such as heavy metal content and salinity, may also affect organic residue decomposition and N mineralization. In a laboratory study, Aoyama and Nagumo (1997) found less C and N released from orchard grass (*Dactylis glomerata* L.) residue in soil treated with 20 mmol Cu kg<sup>-1</sup> than in control soil. Similarly, Ekenler and Tabatabai (2002) found that the addition of metals such as Ag, Hg, and Cu inhibited the activity of  $\beta$ -glucosaminidase, an enzyme involved in N mineralization in soils. Olsen et al. (1996) reported that an increase in the salinity of irrigation water decreased the flux of CO<sub>2</sub> from microcosms treated with plant residues. Clearly, additional research should be conducted in this area to further elucidate the effect of soil characteristics on net N mineralization from organic residues. As in the case of drying and rewetting events, future research should include evaluation of the microbial and microfauna populations involved.

## MODELING NITROGEN MINERALIZATION

### Laboratory Conditions

Laboratory studies of N mineralization have been typically performed under temperature and water content conditions that are optimal or close to optimal for the mineralization process (Quemada and Cabrera, 1995; Gordillo and Cabrera, 1997a). Under such conditions, exponential models (single, or double) have been commonly used to describe cumulative net N mineralized. The single exponential model is of the form:  $N_{\min} = N_o[1 - \exp(-kt)]$ , where  $N_{\min}$  is cumulative net nitrogen mineralized in time  $t$ ,  $N_o$  is potentially mineralizable nitrogen, and  $k$  is the rate constant of mineralization. The double exponential model is of the form  $N_{\min} = N_f[1 - \exp(-kft)] + N_s[1 - \exp(-kst)]$ , where  $N_f$  is a fast pool of mineralizable,  $kf$  is the rate constant of mineralization of the fast pool,  $N_s$  is a slow pool of mineralizable N, and  $ks$  is the rate constant of mineralization of the slow pool.

The single and double exponential models are intended to identify pools of mineralizable N within the organic residue under consideration. In theory, these pools are of defined sizes that should not change with environ-

mental conditions or with the procedure used to fit the models to the data. Research has shown, however, that pools and rate constants in the exponential models are inversely related, which suggests that the same fit to available data could be obtained by increasing one parameter while decreasing the other (Paustian and Bonde, 1987). Research has also shown that increasing the length of incubation used to collect N mineralization data can increase or decrease the pools while decreasing or increasing the rate constants, respectively (Cabrera and Kissel, 1988a; Wang et al., 2003). These problems with exponential models suggest that they could not be used to identify pools of defined, fixed sizes (Sierra, 1990; Dou et al., 1996; Dendooven et al., 1997). To solve this problem, some researchers have proposed fixing the rate constants while allowing the pools to vary to fit the data (Wang et al., 2003). This approach has been used to identify pools of mineralizable N in soil organic matter (Christensen and Olesen, 1998; Wang et al., 2003), and may be useful to identify pools of mineralizable N in organic residues. Preliminary work in this area looks promising. For example, in a study in which 15 poultry litter samples were incubated with soil to measure net N mineralized, the rate constant of mineralization of the slow pool did not vary significantly among poultry litter samples, with an average value of 0.036 d<sup>-1</sup> (Gordillo and Cabrera, 1997a). Thus, it may be possible to use a fixed rate constant for the slow pool. In the same study, the rate constant for the fast pool varied among poultry litter samples, but because the large value of the constant (1.2 d<sup>-1</sup>) indicated that >97% of the fast pool mineralized within the first 3 d, it would seem that, for all practical purposes, an accurate determination of the rate constant would not be as important as an accurate determination of the size of the fast pool. In that study, the fast pool was strongly related to the uric acid content of the litter ( $r = 0.90$ ). Thus, for poultry litter it may be possible to estimate the size of the slow pool of mineralizable N ( $N_s$ ) by fitting the following model to mineralization data:

$$N_{\min} = N_f + N_s[1 - \exp(-0.036t)]$$

where  $N_f$  is the fast pool estimated from the uric acid content of the litter, and  $t$  is time in days. Further research on similar approaches may prove productive with other organic residues.

Another area that warrants further exploration is the procedure used to fit nonlinear models to N mineralization data. Typically, nonlinear regression is used to fit nonlinear models to cumulative net N mineralized with time (Dendooven et al., 1997; Wang et al., 2003). The main problems with fitting cumulative data are that (i) experimental loss of one measurement increment implies complete loss of that replication and (ii) errors from one measurement increment are added to those from subsequent increments, therefore accumulating errors. Fitting incremental data instead of cumulative data reduces the interdependence of observation errors and allows the use of data with missing observations in some of the measurement increments (Ellert and Bettany, 1988;



Hess and Schmidt, 1995). Thus, fitting incremental instead of cumulative data should be further explored.

### Field Conditions

The double and single exponential models, with the rate constants modified according to soil temperature and water content, have been used to estimate net N mineralized from soil organic matter under field conditions (Campbell et al., 1984; Cabrera and Kissel, 1988b). A similar approach has been used to estimate N released from organic residues, particularly for residues that lead to net N mineralization (Gilmour and Skinner, 1999; Gilmour et al., 2003). For organic residues that lead to net N immobilization, however, there are advantages to using more complex simulation models that take into consideration N immobilization by the microbial biomass involved in the decomposition process (Whitmore, 1996; Henriksen and Breland, 1999; Corbeels et al., 1999). A recent review of many of these models can be found in Shaffer et al. (2001). In spite of the fact that these models simulate microbial N immobilization, some studies have shown that simulation models may not accurately simulate N immobilization. For example, Quemada and Cabrera (1997a) found that the N subroutine of the CERES models (CERES-N) did not simulate the initial N immobilization observed during decomposition of wheat (*Triticum aestivum* L.) and oat (*Avena sativa* L.) cover crop residues under field conditions. In recent work, Zubillaga et al. (2004) modified the N requirements of the microbial biomass in CERES-N to simulate the immobilization of N observed after soil incorporation of cotton leaves and stems. Additional work should be conducted in this area to improve the simulation of N immobilization. Furthermore, emphasis should be placed on models that, in addition to simulating N mineralization and immobilization, also consider the dynamics of the microbial population and microfauna present in soil (Fu et al., 2000). These hybrid models that include processes and organisms may be helpful in understanding N cycling in soil and in obtaining better estimates of net N mineralized from organic residues.

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