Crop Residue Management: Effects on Soil Quality, Soil Nitrogen Dynamics, Crop Yield and Nitrogen Recovery

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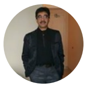

CROP RESIDUES AND MANAGEMENT PRACTICES: EFFECTS ON SOIL QUALITY, SOIL NITROGEN DYNAMICS, CROP YIELD, AND NITROGEN RECOVERY

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This review reveals that crop residues of common cultivated crops are an important resource not only as a source of significant quantities of nutrients for crop production but

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also affecting soil physical, chemical, and biological functions and properties and water and soil quality. When crop residues are returned to the soils, their decomposition can have both positive and negative effects on crop production and the environment. Our aim as agricultural scientists is to increase the positive effects. This can only be achieved with the better understanding of residue, soil, and management factors and their interactions, which affect the decomposition and nutrient release processes. Data on nitrogen benefits and nitrogen recoveries from residues show that a considerable potential exists from residues, especially leguminous residues, not only in meeting the N demands of the succeeding crops, but also in increasing the long-term fertility of the soils. In addition, crop residues and their proper management affects the soil quality either directly or indirectly. Intensive cropping systems are very diverse and complex, so no one residue management system is superior under all situations. Ideally, crop residue management practices should be selected to enhance crop yields with a minimum adverse effect on the environment. It is suggested that in each cropping system, the constraints to production and sustainability should be identified and conceptualized to guide toward the best option. Multidisciplinary and integrated efforts by soil scientists, agronomists, ecologists, environmentalists, and economists are needed to design a system approach for the best choice of crop residue management system to enhance both agricultural productivity and sustainability.

I. INTRODUCTION

Present-day agriculture evolved as we controlled nature to meet our food and fiber needs and to support the increasing population and urbanization of society. In the last thousands of years, the struggle for survival in the seemingly hostile natural environment led to the dominant role of human beings in exploiting soils for increasing food and fiber production. Ancient writings from early civilizations in Mesopotamia, Greece, and India suggested that nutrient supply to the soil, to replenish its lost fertility or to augment its productivity, is a time-honored practice that started after the settled agriculture struck roots.

The unabated use of chemical fertilizers in the last four to five decades has led to the paralleled corresponding decline in the use of cover crops and organic manures (Power and Papendick, 1985). Increased monoculture production of cash grain crops and greater reliance on the importation of chemical fertilizers and pesticides to maintain crop growth have resulted in greatly increased grain yields and labor efficiency. However, these conventional management practices have led to the decline in soil organic matter (SOM), increased soil erosion, and surface and groundwater contamination (Reganold *et al.*, 1987). Until recently, we failed to recognize the consequences of management on the balances and cycling of energy and matter and soil productivity (Goh and Nguyen, 1992). Awareness of the environmental aspects of soil quality and crop production has been increasing in recent years, which has led to the renewed interest in crop residues, green manures, and other organic manures as sources of SOM and nutrients for crops.

Crop residues are a tremendous natural resource—not a waste. Residue management is receiving a great deal of attention because of its diverse effects on soil physical, chemical, and biological properties. The quantities of nutrients that can be returned annually to soils as residues of common cultivated crops are considerable, requiring worthwhile consideration. Biological nitrogen fixation (BNF) by leguminous crops and the recycling of fixed N when leguminous crop residues are returned to the soil can be a significant source of N to the soil organic N pool as well as for subsequent plant uptake. The amount of N that recycles into agricultural fields through residues may add 25–100 Tg of N year^{-1} into agricultural soils (Mosier and Kroeze, 1998). Thus, there is a need to determine sink sizes and turnover rates of different quality residues and to increase the efficiency of nutrient cycling from residues through different soil sinks, and eventually to growing plants, with minimum loss from the system.

This may involve development of practices to enhance the immobilization of nutrients when plants are not growing, as well as practices to increase nutrient availability when plants are actively growing. Information on the kinetics of decomposition of the crop residues and mineralization–immobilization turnover of different quality residues (leguminous and nonleguminous) is required to ascertain the actual amount of crop residues needed to maintain the soil productivity and to ensure environmental protection by minimizing nutrient losses and soil erosion. Addition of organic matter (OM) to the soil through the return of crop residues also improves soil structure, influences soil water, air, and temperature relations, helps control runoff and erosion, and makes tillage easier.

II. CROP RESIDUES AND THEIR USES

A. Definition of Crop Residues

Crop residues, in general, are parts of the plants left in the field after crops have been harvested and thrashed or left after pastures are grazed. These materials have at times been regarded as waste materials that require disposal but it has become increasingly realized that they are important natural resources and not wastes.

B. World Crop Residue Production and Utilization

According to Lal (1995) in the United States, annual crop residues produced by 19 principal crops are estimated as 400 million tons year^{-1}, compared with 2962 million tons year^{-1} produced in the world. The total amount of principal nutrients in crop residues ranges from 40 to 100 kg tons^{-1}. Major nutrients (N + P + K) contained in crop residues amount to 9 million tons year^{-1} in the United States and 74 million tons year^{-1} in the world. Although large amounts of nutrients are removed in harvested grain, significant amounts are still contained in the plant residues after harvest (Table I). The management of crop residues has important implications for the total amounts of nutrients removed from the soil. A considerable quantity of fertilizers can be saved by returning crop residues produced to the soil. In addition, approximately 1.5 Pg (1 Pg = 10^{15} g) of carbon (C) is stored in the crop residues produced in the world, which can be an important source of OM added to the soils. Crop residues can be used for improving soil health and productivity and are a major source of lignocellulose entering the soil. Lignocellulolysis can have positive or negative effects on crop productivity, and the challenge is to enhance the positive value of plant residue decomposition at the expense of the negative value (Smith *et al.*, 1992). In developing countries, most crop residues are used as animal feed and housing material and in industrial uses such as paper making (Stumborg *et al.*, 1996; Latham, 1997; Powell and Unger, 1997). Only a part of crop residues is used as soil amendment.

III. DECOMPOSITION OF CROP RESIDUES

The decomposition of crop residues is a microbial-mediated progressive breakdown of organic materials with ultimate end products C and nutrients released into the biological circulation in the ecosystem at both a local and a global scale. Crop

Table I

Amounts of Chemical Constituents Found in 1 Tonne of Residues of Three Cereal Straws, Maize Residues, and Pasture Herbage[a]

	Approximate content (kg ton^{-1} residue)				
	Barley	Oats	Wheat	Pasture	Maize
Nitrogen	4.6	5.9	6.9	21.2	8.4
Phosphorous	0.4	0.6	0.8	2.7	2.0
Potassium	14.3	23.3	13.5	24.6	16.5
Sulfur	1.4	1.1	1.3	2.0	nd[b]
Calcium	2.6	1.4	1.8	5.6	nd
Magnesium	0.8	0.5	0.8	1.4	nd

[a] From Fraser and Francis (1996).

[b] Not determined.

residues represent only a fraction of total C in soil but these decompose very rapidly. It has been estimated that about half of the global CO_2 output from soil originates from the decomposition of annual litter fall (Coûteaux *et al.,* 1995). However, there is a vast pool of stable OM in the soil, but this decomposes very slowly—over centuries or millennia (Campbell *et al.,* 1967; Jenkinson and Rayner, 1977; Goh *et al.,* 1984; Parton *et al.,* 1987).

Coûteaux *et al.* (1995) stated that plant residue decomposition involves two simultaneous and fundamental processes: the concomitant mineralization and humification of C compounds by microorganisms and the leaching downward in the soil of soluble compounds, whose C and N are progressively mineralized and immobilized.

According to Gregorich and Janzen (1998), in natural ecosystems the decomposition synchronizes with plant growth and C and other nutrients are utilized in the system with maximum efficiency. However, disturbance of these ecosystems may retard or accelerate the decomposition process relative to the other ecosystem processes and may lead to the deterioration of some of the components of the ecosystems. An understanding of this process will help to ensure the proper management of this important resource.

A. Factors Affecting Crop Residue Decomposition

Residue decomposition processes are controlled by three main factors: (i) kind of plant residues, (ii) edaphic factors, and (iii) residue management factors. Edaphic factors are dominant in areas subjected to unfavorable weather conditions, whereas plant residue factors largely play a role as regulator under favorable environmental conditions. Many of these factors are not independent as a change in one factor may affect other factors. For example, high soil moisture may result in lower soil temperature and aeration and surface residue application may affect soil moisture and temperature simultaneously. Because of these strong interactions, it is often difficult to isolate the effects of specific environmental factors on residue decomposition.

1. Crop Residue Factors

a. Residue Particle Size

Ground plant material has often been used for convenience in the study of plant residue decomposition because of their uniform substrate. There is controversy regarding the effect of plant residue particle size on the rate of residue decomposition, and mineralization–immobilization turnover (MIT) of N in the soil (Angers and Recous, 1997; Jensen, 1994b; Sorensen *et al.,* 1996). Small particles may decompose faster than larger particles because of the increased surface area and greater dispersion in soil increasing the susceptibility to microbial attack due to lack of lignified barrier tissue (Summerell and Burgess, 1989), especially if residues are not penetrated readily by fungi and bacteria (Amato *et al.,* 1984; Jensen, 1994b; Nelson *et al.,* 1996; Angers and Recous, 1997). Although soil fauna are responsible for only a small proportion (<10 %) of soil respiration (Juma and McGill, 1986; Anderson, 1991), they play an important role in increasing the rate of decomposition by comminuting and redistributing OM, making it more accessible to microbial attack. For example, Curry and Byrne (1992) reported a 26–47% faster decomposition of straw in mesh bags allowing earthworms than in bags that excluded earthworms during a 8- to 10-month period of decomposition. However, the microbial biomass and products formed during the initial decomposition of small particles may be better protected against further decomposition due to a more intimate mixing of the mineral soil (Van Schrevan, 1964; Saggar *et al.,* 1996; Skene *et al.,* 1996).

In addition, an enhanced exposure to nonsoluble carbohydrates materials can lead to N immobilization (Van Schrevan, 1964), and an enhanced surface area can lead to exposure to more phenolic substances that are known to inhibit decomposition (Vallis and Jones, 1973; Fox *et al.,* 1990). Ambus and Jensen (1997) reported that the higher microbial activity associated with the initial decomposition of ground plant material was due to a more intimate plant residue–soil contact, but in the long term, grinding of plant residues had no significant effect on N dynamics. The effect of plant residue particle size on MIT may thus be an interaction between clay and silt content, secondary metabolic products, plant residue chemical composition, period of decomposition, and faunal activity. The management of residue particle size and the degree of mechanical destruction may thus be important for the conservation of N in agricultural systems.

b. Age of Residue

The chemical composition of most crop plants changes dramatically during their growth period (Luna-Orea *et al.,* 1996). As the plant matures, its content of protein and water-soluble constituents decreases steadily, whereas the amount of hemicellulose, cellulose, and lignin increases.

In general, water-soluble fractions (sugars, organic acids, proteins, and part of structural carbohydrates) are degraded first (Reber and Scharer, 1971; Knapp *et al.,* 1983a,b) followed by structural polysaccharides (cellulose and hemicellulose) (Harper and Lynch, 1981) and then lignin (Harman *et al.,* 1977; Stout *et al.,* 1981; Collins *et al.,* 1990b). Consequently, the residue of immature plants generally decomposes more readily than those of older plants (Wise and Schaefar, 1994; Cortez *et al.,* 1996) and, as a result, releases more nutrients (Luna-Orea *et al.,* 1996).

c. Leaf Toughness

Physical leaf toughness affecting residue decomposition has received little attention. Gallardo and Merino (1993) developed a toughness index of residue and proposed leaf toughness as an index of substrate quality. Silica content is responsible for leaf toughness and has been reported to affect the digestibility of plant material and their decomposition (Goering and Van Soest, 1970; Ma and Takahashi, 1989). In general, the greater the silica content, the slower the decomposition.

d. Desiccation

The drying of crop residues before incorporation is a common procedure in N mineralization studies. Heat drying plant materials even at low temperatures between 50 and 60°C can produce analytically significant increases in lignin concentration because of the production of artifact lignin via a nonenzymatic browning reaction that involves plant N (Goering and Van Soest, 1970; Moore *et al.*, 1988). This resulted in a significant reduction in N mineralization from the residues compared to fresh or freeze-dried residues (Moore *et al.*, 1988). Likewise, greater N mineralized from fresh compared to freeze-dried clover residues has been reported (Breland, 1994).

2. Crop Residue Quality

Plants contain 15–60% cellulose, 10–30% hemicellulose, 5–30% lignin, 2–15% protein, and soluble substances, such as sugars, amino acids, amino sugars, and organic acids, which may contribute 10% of dry weight (Paul and Clark, 1989). Plants also contain cutin (Gallardo and Merino, 1993), polyphenols (Tian *et al.*, 1995b), and silica (Goering and Van Soest, 1970). The rate of organic matter breakdown depends on the relative proportions of each of these fractions, such as soluble sugars, cellulose, hemicellulose, and lignin (Stout *et al.*, 1981). Hagin and Amberger (1974) reported that the half-lives of sugars, hemicellulose, cellulose, and lignin were 0.6, 6.7, 14.0, and 364.5 days, respectively. It has long been recognized that the fractional loss rate declines with time (Minderman, 1968; Jenkinson, 1977; Mellilo *et al.*, 1989; Andren *et al.*, 1990; Bending *et al.*, 1998), and this decline reflects the decline in the quality of the remaining substrate.

a. C/N Ratio and Nitrogen Content

Crop residues contain about 40–50% C on dry weight basis, but their N content varies considerably, causing the variation in C/N ratios. It is generally accepted that residues with a wide C/N ratio decompose more slowly than those with a narrow C/N ratio (Parr and Papendick, 1978), and plant residues with high N content show high decomposition rates and nutrient release (Janzen and Kucey, 1988; Douglas and Rickman, 1992). Highly significant correlations among N content, N release, and biomass loss have been reported by many workers (Frankenberger and Abdelmagid, 1985; Mellilo *et al.*, 1982; Neely *et al.*, 1991; Giller and Cadisch, 1997). Other studies have also reported the importance of initial N content for determining residue decomposition (Aber and Mellilo, 1980; Berendse *et al.,* 1987; Janzen and Kucey, 1988; Vigil and Kissel, 1991). A high N content of residues reduces competition of available N by microorganisms and consequently enhancing the decomposition by maintaining high microbial activity. Bangar and Patil (1980) noted that an addition of N to lower the C/N ratio of wheat straw (75:1) significantly resulted in liberation of more CO_2 than control. Oh (1979) also reported an enhanced rate of crop residue decomposition on the addition of farmyard manure.

Christensen (1986) showed that 44% of straw (0.92 % N) decomposed during the first month but that only 7% of the straw (containing 0.4% N) decomposed during the same period of incubation. DeHaan (1977) found no correlation between percentage N of added plant tissue and the rate of decomposition. Jensen (1989) noted that green tops and fresh roots of pea plants (3.9% N) decomposed quickly. Douglas *et al.* (1980) reported 15% weight loss in 60 days for wheat residues with initial N contents <5.5 g kg^{-1} and 30% when N contents were >5.5 g kg^{-1}, whereas Reinstern *et al.* (1984) reported that 1.13% N straw decomposed 2.3 and 1.6 times faster than 0.18 and 0.79 % N straws, respectively.

The threshold C/N ratio, above which decomposition is suppressed, is often about 20 to 30. However, C/N ratios and N contents have not always correlated well with decomposition rates and better explanations are needed. Reinstern *et al.* (1984) postulated from their studies using extraction and leached straw samples that microbial biomass production and wheat straw decomposition rates in the early stages were largely dependent on the size of the water-soluble C pool. Crop residue decomposition based on available C and N seems to relate more closely to field observations than decomposition based on total C and N. Available C for microbial decomposition has been estimated for different plant residues, which correlated with decomposition (Mtambanengwe and Kirchmann, 1995).

Although the N content and C/N ratio of crop residues are useful in predicting residue decomposition rates, these should be used with some caution as the C/N ratio reveals little on the availability of C and N to microorganisms. Any factor that increases the rate of decomposition, and hence the N demand, tends to increase the threshold N concentration (lower the threshold C/N ratio). For example, a more favorable climate and higher rates of residue application with a greater amount of readily available C in the substrate would stimulate greater microbial activity, increase N demand, and increase the threshold N concentration. The information generated from laboratory studies conducted in a more favorable environment would therefore provide misleading estimates of threshold C/N ratios or may even overstate the impact of N content on field residue decomposition rates (Dendooven *et al.,* 1990).

b. Lignin

The role of lignin as an inhibitor in the decomposition process has been elucidated in several studies (Meentemeyer, 1978; Berendse *et al.,* 1987; Fox *et al.,* 1990; Vanlauwe *et al.,* 1996; Hammel, 1997; Giller and Cadisch, 1997). Lignin is known to be a recalcitrant substance, highly resistant to microbial decomposition (Mellilo *et al.,* 1982), and only relatively few microorganisms can degrade lignin and these are exclusively aerobic (Jenkinson, 1988). Many workers have found that increasing the lignin concentration reduces the decomposition rate and nutrient release from plant residues and also enhances nutrient immobilization, especially N (Aber and Mellilo, 1982; Aber *et al.,* 1990; Tian *et al.,* 1992a). Fogel and Cromack (1977) have shown that the lignin concentration of the substrate was an excellent index for predicting rates of decomposition and weight losses of forest litter samples. Muller *et al.* (1988) and Rutigliano *et al.* (1996) found that the lignin concentration was a much better predictor of the residue decomposition rate than N concentration.

c. Polyphenols

Vallis and Jones (1973) suggested that polyphenols bind to protein and form complexes resistant to decomposition. Polyphenols can also bind to organic N compounds (amino acids and proteins) in leaves, making N unavailable, or bind to soluble organic N released from leaves, forming resistant complexes in the soil (Northup *et al.,* 1995). Polyphenols also inhibit enzyme action (Swain, 1979). Sivapalan *et al.* (1985) found lower net N mineralization from tea leaves with high soluble N and high polyphenol content in comparison with those with high soluble N but low polyphenol content. Jensen (1989) reported that the top growth of legumes was among the most rapidly degradable plant materials because of being high in protein and low in lignin and other inhibitors such as polyphenol compounds.

The importance of polyphenols in residue decomposition and the mineralization process has been debated frequently (Swift *et al.,* 1979). In some studies, ployphenols and ployphenol/N and (lignin + polyphenol)/N ratios have been correlated with residue decomposition and nutrient release, whereas in other studies, N content, lignin content, and lignin/N ratios were better correlated with residue decomposition and N release (Haynes, 1986; Fox *et al.,* 1990; Palm and Sanchez, 1991; Vigil and Kissel, 1991; Thomas and Asakawa, 1993; Constantinides and Fownes, 1994a).

d. Combined Chemical Composition

Herman *et al.* (1977) and Tian *et al.* (1995b) found that the decomposition rate of plant residues could not be predicted from individual property of the organic material such as C/N ratio, lignin content, or carbohydrate content, but when combined these properties could accurately predict relative rates of decomposition for a broad range of plant residues.

According to Berg and Agren (1984) and Janzen and Kucey (1988), residue decomposition occurred in two phases. Phase I is relatively rapid and is dependent on the initial residue N content (Jama and Nair, 1996), whereas phase II decomposition is relatively slower and is regulated by lignin and polyphenol decomposition (Berg, 1986; Jama and Nair, 1996), which shows little differences in the residue decomposition rate regardless of initial N content because soluble, easily decomposable components have already been utilized by microbes or lost by leaching (Reinstern *et al.,* 1984; Christensen, 1986; Smith and Peckenpaugh, 1986; Collins *et al.,* 1990a; Douglas *et al.,* 1990). Summerell and Burgess (1989) found no relationship between the amount of a chemical component and the rate of straw decomposition other than the higher percentage of water-soluble compounds in barley straw, which may be due to phase II decomposition. The distinction between these two phases occurred when the "lignocellulose index" [LCI = ratio of lignin/(lignin + cellulose)] reached a value of about 0.7 (Mellilo *et al.,* 1989). When this LCI value has been reached, the composition of decaying material remained unchanged and the decay was determined by environmental factors. Both C and N dynamics were broadly described by this two-phase model (Mellilo *et al.,* 1989).

These sequential patterns of residue utilization can result in a shift in the relative variable controlling decomposition and nutrient mineralization (Mellilo *et al.,* 1989). Berg and Staff (1980) showed a shift from nutrient and soluble C control in the early stages of decomposition of *Pinus sylvestris* needles to the dominance of lignin as the controlling factor in later stages. Similarly, it has been shown that the polyphenol/N ratio may serve as a short-term index for green manures, whereas (lignin + polyphenol)/N provides an index of long-term release for more woody and naturally senescent material (Palm, 1995).

Microbial, particularly fungal, succession on decomposing litter reflects changes in litter composition, as do fauna with recognition of phases in palatability and interaction with microflora (e.g., Scheu and Wolters, 1991; Van Wensem *et al.,* 1993; Hammel, 1997). A consequence of this is that correlations between rate of mineralization or nutrient loss and simple expressions of the initial composition of litter will have limitations (Heal *et al.,* 1997). These changes are not only restricted to chemistry. For example, Gallardo and Merino (1993) distinguished an initial leaching phase in which the leaf toughness and toughness-to-P concentration of the original litter provided the best prediction of mass loss; in contrast, the cutin-to-N ratio and cutin concentration were the best predictors in the postleaching phase (Palm, 1995).

3. Methodological Problems Associated with Residue Quality Characterization

In recent years, several attempts have been made to quantify residue quality and its relationships with residue decomposition, mostly in terms of N mineralization (Palm and Sanchez, 1991; Oglesby and Fownes, 1992; Kachaka *et al.,* 1993; Bending *et al.*, 1998). Although general trends have been observed, no unique relationship has been developed (Vanlauwe *et al.*, 1997). This is partly due to different methodologies and approaches used by different workers.

a. Extraction Methods for Lignin and Polyphenols

Different methods are used for extracting polyphenols from plant tissues. Amounts of total ployphenols extracted from plant tissues varied from 30 to 90% according to a method used (Swain, 1979; Anderson and Ingram, 1989; Quarmby and Allen, 1989; Constantinides and Fownes, 1994b; Vanlauwe *et al.*, 1997). Tedious and inaccurate methods of proximate analysis obscured the biochemical composition. The advent of more sophisticated techniques (e.g., variants of mass spectrometry) allows the rapid and sensitive characterization of organic materials, thus enabling their degradation and synthesis to be followed (Sanger *et al.*, 1996; Heal *et al.*, 1997).

b. Age of Plant Residues and Molecular Size of Polyphenols

The concentration of polyphenol is generally greater in mature residues than in green leaves (Fox *et al.*, 1990; Palm and Sanchez, 1991; Thomas and Asakawa, 1993), and polyphenols have different properties with respect to binding N-containing compounds, depending on their molecular weights (Scalbert, 1991). These explained why ployphenols correlated with decomposition and N release in some studies but not in others (Fog, 1988; Fox *et al.*, 1990; Palm and Sanchez, 1991; Thomas and Asakawa, 1993; Vanlauwe *et al.*, 1996).

c. Methods of Determining Decomposition Rates

Different particle sizes of crop residues and methods of determining residue decomposition (viz. direct application to soil or application in mesh bags) are known to affect residue decomposition rates (Summerell and Burgess, 1989; Fox *et al.*, 1990; Constantinides and Fownes, 1994a; Magid *et al.*, 1997a). Variations in residue weight loss determinations using mesh bags, which is the most common method of estimating decomposition, will be discussed later (Section IIIB).

d. Variation in Composition of Same Plant Species at Different Sites and Different Plant Parts

Several workers have reported differences in residue decomposition due to differences in N, C/N, lignin/N, and polyphenol/N ratios even for the same species. Variation of site, plant part, and environment conditions also affected the litter chemistry of residues from the same plant species (Harper and Lynch, 1981; Berg and Tammy, 1991). For example, *Pinus sylvestris* needles varied in carbohydrate composition and lignin polymerization in relation to nutrient status and pH of different soils on which they were grown (Sanger *et al.*, 1996). Likewise, Vitrusek *et al.* (1994) found that the litter of tropical tree *Metrosideros polymorpha* grown on dry Hawaiian lava flows decomposed twice as rapidly as litter of the same species on wet sites. These workers concluded that higher substrate quality from dry sites could be due to trade-offs in nutrient and water use efficiency and C gain by plants when grown under different climates.

An unconventional approach was adopted by Cornelissen (1996) using the strategy theory of Grime (1988), where the weight loss of more than 100 species of leaf, needles, and shoot litter was measured under standard field conditions. The weight loss was related to a variety of plant characteristics such as growth habits, evergreen vs deciduous, autumn correlations, and evolutionary advancement. There was a clear evidence of the adaptive strategy of tissues defense (accumulate different chemicals in their tissues). Cornelissen's (1996) results suggested that environmentally stressed habitats produced relatively slow decomposing leaves.

4. Residue Decomposition Index/Quality Index

Attempts have been made to predict the rate and pattern of decomposition of organic substrates from a range of organic materials based on their chemical components (e.g., C/N ratio, lignin content, lignin-to-N ratio, polyphenol). Herman *et al.* (1977) proposed the decomposability index as

$$\text{Decomposability index} = \frac{(\text{C/N}) \times (\%\ \text{lignin})}{\sqrt{(\%\ \text{carbohydrates})}} \qquad (1)$$

It was found that this decomposition index correlated inversely with total CO_2 evolution from the decomposing roots of three grass species (Herman *et al.*, 1977) and predicted accurately the decomposition of other plant materials, including legume residues.

Cortez *et al.* (1996) found that parameters integrating lignin were highly correlated to the decomposition of a wide variety of litter. These workers developed the HLQ index as

$$\text{HLQ} = \frac{\text{hemicellulose} + \text{cellulose}}{\text{hemicellulose} + \text{cellulose} + \text{lignin}} \qquad (2)$$

Tian *et al.* (1995b) developed the plant residue quality index (PRQI) for assessing the quality of plant residues as follows:

$$\text{PRQI} = \frac{1}{(a \times \text{C/N ratio} + b \times \text{lignin} + c \times \text{polyphenols})} \times 100, \qquad (3)$$

where *a, b,* and *c* are coefficients of relative contribution of C/N ratio, lignin content (%), and polyphenol content (%) to plant residue quality.

The PRQI was found to be correlated with the decomposition rate of plant residues using litter bags. These workers concluded that PRQI can be used for selecting plant residue and projecting their agronomic value.

Janssen (1996) proposed a resistance index (RI) that depends on the decomposability of different residues. It was found from a desk study using some of the earlier published work that good linear relationships existed between the fraction of organic N mineralized and initial C/N ratio of the substrate for organic materials with similar decomposability.

It is thus obvious that a combination of lignin and polyphenol concentration offers perspectives for the quantitative evaluation of decomposability. However, this needs further evaluation on a wide variety of organic materials with a standard proximate analysis of lignin and polyphenol concentration before a universal plant residue quality index could be developed.

5. Edaphic Factors

a. Soil pH

Soil pH is one of the most important factors influencing residue decomposition as it affects both the nature and size of population of microorganisms and the multiplicity of enzymes at the microbial level, which subsequently affect decomposition (Paul and Clark, 1989). In general, the decomposition of crop residues proceeds more rapidly in neutral than in acid soils. Consequently, liming acid soils accelerate the decay of plant tissues, simple carbonaceous compounds, and SOM (Alexander, 1977; Condron *et al.,* 1993). Under field conditions in the United Kingdom, Jenkinson (1977) reported that 42% of the ryegrass-derived C still remained after 1 year in a soil of pH 3.7, whereas only 31% remained in soils of pH between 4.4 and 6.9. This may be due to alterations in soil microbial populations and activity as soil pH changes. Characteristically, the population shifts from bacteria to actinomycete to fungi as soil pH declines (Alexander, 1980).

b. Soil Temperature

Parr and Papendick (1978) and Stott *et al.* (1986) reported that microbial decomposition processes are more important than physical and chemical processes in causing the loss of residues from the field, thus releasing nutrients. Temperature affects the physiological reaction rates of organisms and the activity of microbial cells by the laws of thermodynamics and hence microbial activity (Paul and Clark, 1989) and residue decomposition (Westcott and Mikkelson, 1987; De-Neve *et al.,* 1996).

The influence of temperature on crop residue decomposition has been described quantitatively as the temperature quotient Q_{10}. Values of Q_{10} for the N mineralization rate of native SOM in the temperature range between 5 and 35°C have been reported to be approximately 2 (Stanford *et al.,* 1975; Campbell *et al.,* 1981; Kladivko and Keeney, 1987; Scholes *et al.,* 1994). Higher Q_{10} values have been reported by other workers between different temperature ranges (Pal *et al.,* 1975; Addiscot, 1983; Campbell *et al.,* 1984; Vigil and Kissel, 1995), indicating some interactions between temperature and quality of crop residues.

Microorganisms function at maximum growth and activity in the temperature range of 20–40°C and showed maximum decomposition in this range (Roper, 1985; Stott and Martin, 1989). However, significant straw decomposition can occur even at 0°C (Stott *et al.,* 1986, 1990; Kanal, 1995). At the extreme ends of the temperature scale (e.g., 0 and 40°C), it is generally believed that temperature regulates the activity of microorganisms more than their mass.

Under field conditions, marked diurnal and seasonal fluctuations in surface soil temperature are common (Biederbeck and Campbell, 1973). Although it has generally been found that microbial growth is inhibited by fluctuating temperatures (Biederbeck and Campbell, 1971), some studies showed that N mineralization remains virtually unaffected in the mesophilic (15–45°C but optimum between 25 and 35°C) temperature range (Stanford *et al.,* 1975).

c. Soil Moisture

The growth and activity of soil microorganisms rely on soil moisture, which, in turn, produces significant effects on plant residue decomposition and nutrient cycling (Stanford and Epstein, 1974; Sommers *et al.,* 1981; Schomberg *et al.,* 1994). Sommers *et al.* (1981) observed that soil dried to a water potential of -10 Mpa evolved CO_2 at about half the rate of soils incubated at optimal water content (-20 to -50 kPa).

Pal and Broadbent (1975) showed that the maximum rate of decomposition for plant residues occurred at 60% water holding capacity (WHC) and the rates decreased at either 30 or 150% of WHC. Summerell and Burgess (1989) reported that the rate of straw decomposition as measured by dry weight loss was highest at -0.1 MPa and decreased as the external soil water potential was lowered.

Das *et al.* (1993) observed significantly more N release from decomposing crop residues at field capacity than at 50% of field capacity. Thomsen (1993) reported more soil microbial biomass on straw addition under moist soil conditions (54–82%) than under wet conditions (4–27%), probably due to limited aeration for microbial activity under wet conditions. Thus, both very dry and very wet conditions of soil inhibit decomposition by limiting either moisture content or soil aeration for microbial activity.

d. Freezing and Thawing

The thawing of previously frozen surface detritus resulted in the immediate release of large amounts of soluble materials (Witkamp, 1969; Bunnell *et al.,* 1975). This is thought to represent the release of materials previously immobilized in microbial tissue (Witkamp, 1969). Such a release of soluble materials contributed to the burst of decomposer activity that occurred at the onset of snow melt (Bunnell *et al.,* 1975), which may lead to a substantial increase in the decomposition rate.

e. Drying and Rewetting

The effect of drying and rewetting on the decomposition of plant residues is unclear. For example, Van Schrevan (1968) found that although drying stimulated the subsequent mineralization of C and N from soil humus, it retarded the mineralization of fresh plant materials. In another study, soil drying and wetting were found to promote the turnover of C derived from added ^{14}C-labeled plant material, and the increase in C was mainly due to an enhanced turnover of microbial products (van Gestel *et al.,* 1993). The decay rate of biomass ^{14}C increased relatively greater by soil desiccation and remoistening than decay rates of nonbiomass ^{14}C. Haider and Martin (1981) found that drying and rewetting produced no effect on the decomposition of ^{14}C-labeled lignin when incorporated into soil, but the decomposition of added cellulose in soils was found to increase (Sorensen, 1974).

Repeated drying and wetting of the soil appeared to increase the resistance of certain N compounds of the plant to microbial decomposition. Franzluebbers *et al.* (1994) reported that repeated drying and rewetting did not reduce the C mineralization of cowpea [*Vigna unguiculata* (L.) Walp.] significantly; N mineralization from cowpea, however, was reduced significantly. Repeatedly drying and wetting can inhibit microbial growth and/or activity severely. In the field, it could reduce N mineralization from legume green manure compared to decomposition in continuously moist soil. This may contribute to long-term soil N fertility by increasing the soil organic N content.

f. Aerobic and Anaerobic Conditions

Decomposition and mineralization are slower and less complete under anaerobic than aerobic conditions (Pal and Broadbent, 1975; Murthy *et al.,* 1991; Kretzschmar and Ladd, 1993). When soils become so wet that larger pores are filled with water, the decomposition of OM is limited by the rate at which oxygen can diffuse to the site of microbial activity, as the diffusion coefficient of oxygen in water is 10,000 times slower than in air. Thus, even a modest oxygen demand cannot be met if larger soil pores are filled with water (Jenkinson, 1988). Reddy *et al.* (1980) showed that the first-order rate constant for rice straw decomposition was 0.0054 day^{-1} for phase I (easily decomposable fraction) and 0.0013 day^{-1} for phase II (slowly decomposable fraction) under aerobic conditions, and corresponding values for anaerobic conditions were 0.0024 and 0.0003 day^{-1}, respectively.

g. Soil Salinity

This is generally attributed to a direct influence of the osmotic potential on microbial activity (Johnston and Guenzi, 1963; Singh *et al.,* 1969) or through the alteration of pH, soil structure, aeration, and other factors (Nelson *et al.,* 1996). Results showed that plant residue composition, as well as increased salinity, affected the decomposition and CO_2 surface flux dissolved organic C and may be an important factor for C storage in saline systems (Hemminga *et al.,* 1991; Hemminga and Buth, 1991; Olsen *et al.,* 1996). In the absence of pH and aeration effects, Nelson *et al.* (1996) concluded that sodicity increased and salinity decreased the decomposition of finely ground plant residues, with no significant interaction.

h. Available Nutrients

The primary limiting factor in most soils for microbial growth is C. An abundance of C is added to the soil through crop residues. New cells of microorganisms require not only C, H, O, and N, but also P, K, S, Mg, Ca, and micronutrients. When an organic material is added to the soil, microorganisms decomposing it can obtain the necessary inorganic nutrients (N, P, K, S, Ca, etc.) for growth from two sources: (i) those already present in the soil in plant available forms and (ii) those in the added organic material itself.

However, Jenkinson (1981) stated that it is unusual for nutrient elements other than N to limit the decomposition of plant or animal material in soil, even though they can restrict microbial activity *in vitro*. It has been widely recognized that the element required in the greatest abundance is N. Nitrogen is required by all decomposers because it is a constituent of extracellular and intracellular enzymes, nucleic acids, and lipoprotein membranes, thus making it the most limiting nutrient for microbial activity. However, the effects of added N on the decomposition of OM are variable (Jenkinson, 1981) and will be discussed in detail later.

Apart from N, other elements have also been shown to influence residue decomposition. For example, Cheshire and Chapman (1996) concluded from adding N and P to ^{14}C-labeled ryegrass that P, whether intrinsic or added, increased the rate of decomposition of organic residues but there was a strong interaction with N, which had a predominant influence. In addition, Enriquez *et al.* (1993) demonstrated that the decomposition rate increased with both intrinsic N and P. The effect of N depends on its form. Increased intrinsic tissue N increased the rate of C loss, whereas added inorganic N decreased the rate of C loss during decomposition (Cheshire and Chapman, 1996).

i. Inorganic Nitrogen

Swift *et al.* (1979) showed that inorganic native N in soil enhanced the mineralization of OM. Woods *et al.* (1987) reported that the application of NH_4^+ under N-limited conditions increased microbial respiration, microbial population, and N mineralization. Negative or no effects of added N on decomposition and microbial activity have also been reported (Fog, 1988; Hassink, 1994a; Cheshire and Chapman, 1996) and have been attributed to (a) the outcome affecting the competition between potent and less potent decomposers through "ammonia metabolite repression" (Keyser *et al.,* 1978); (b) N blocking the production of certain enzymes and enhancing the breakdown of the more available cellulose (Lueken *et al.,* 1962), whereby recalcitrant lignocellulose accumulated and amino compounds condensed with polyphenols; and (c) ammonia toxicity (Vines and Wedding, 1960; Clay *et al.,* 1990).

Thus N fertilizer has the potential to influence residue decomposition and mineralization and types of N compounds found in the soil. In soils with low pH, N mineralization rates of plant residues were reduced when N fertilizer was applied to soils (Clay and Clapp, 1990). Soils with a low pH had reduced nitrification rates resulting in NH_4^+ concentration. The N fertilizer application increased nonhydrolyzable N and decreased the residue-derived amino acid N when compared to unfertilized treatment. Clay and Clapp (1990) explained these results as due to (a) N fertilizer influencing the mineralization rate of different substrates, with some depolymerization products reacting with amino compounds, reducing hydrolyzable amino acid N to form condensation products and resulting in increased amounts of nonhydrolyzable N, and (b) NH_4^+ inhibition of the microbial population. Cerri and Jenkinson (1981) demonstrated a lesser mineralization of plant residue added to a soil rich in easily decomposable OM.

j. Soil Texture, Clay Content, and Soil Structure

Jenkinson (1977) and Ladd and Foster (1988) showed that the decomposition of plant materials was more rapid in soil with less clay content because the clay protected the OM from decomposition. As clay content increases, soil surface area also increases, which results in an increased SOM stabilization potential (Sorenson, 1981; Merckx *et al.,* 1985; Jenkinson, 1988; Ladd *et al.,* 1996; Saggar *et al.,* 1996). This stabilized OM has turnover time ranging from 10 to 1000 years (Parton *et al.,* 1987). The role of clay in stabilizing OM appears to be more important in warmer soils where higher decomposition rates can be expected. In cold soils, cool temperature may be the main factor slowing decomposition, and clay content may be less important (Anderson, 1995). However, comparing different substrates, Skene *et al.* (1997), showed that for high-quality substrates, physical protection by inorganic matrices was a major limiting factor to decomposition, whereas for low-quality substrates, chemical protection is the major limiting factor.

Texture also influences the soil physical environment (Elliott *et al.,* 1980; Hassink *et al.,* 1993), which further affects the microbial activity (Stott *et al.,* 1986; Skopp *et al.,* 1990; Hassink *et al.,* 1993; Killham *et al.,* 1993). Other effects of soil texture may be through N and P availability (Mackay *et al.,* 1987) by influencing total OM accumulation or microbial activity (Schimel *et al.,* 1985; Hassink *et al.,* 1993).

Soil structure also exerts a dominant control over the stabilization of SOM (Van Veen and Kuikman, 1990; Ladd *et al.,* 1996) and OM is protected in microaggregates (Tisdall and Oades, 1982; Skjemstad *et al.,* 1993; Golchin *et al.,* 1994). The degree of physical protection of ^{14}C-labeled residues against decomposition in the microaggregate fraction (<20 mm) was negatively correlated with the degree of saturation of this particle-size fraction with SOM (Hassink and Dalenberg, 1996).

The higher the level of elements retained in the soil, the greater they can be taken up by plants. Jordan *et al.* (1996) found greater residue N uptake by sorghum and soil microbial biomass C and N in clayey soil (34% clay) compared with sandy soil (5% clay).

k. Indigenous Macro- and Microorganisms

Decomposition and mineralization are largely biological processes, mainly accounted for by the activity of the soil microflora. However, abiotic processes cannot be discounted entirely, especially in harsh environments, where appreciable plant litter mass may be lost by abiotic mechanisms, such as fragmentation, physical abrasion, photochemical breakdown, and leaching (Moorhead and Reynolds, 1989; Dormaar, 1991; Coûteaux *et al.,* 1995). Even when these abiotic processes of mass loss predominate, the eventual oxidation to CO_2 and other inorganic constituents is probably mediated by soil organisms. Decomposer organisms generally consist of a complex community of soil biota, including microflora and soil fauna. Fungi and bacteria are ultimately responsible for the biochemical processes in the decomposition of organic residues (Juma and McGill, 1986). Soil fauna enhance the biodegradation and humification of organic residues in several ways (Tian *et al.,* 1997), such as by (a) comminuting organic residues and increasing surface area for microbial activity, (b) producing enzymes that break down complex biomolecules to simple compounds and polymerize compounds to form humus, and (c) improving the environment for microbial growth and interactions.

The direct contribution of the soil mesofauna to these processes, derived primarily from their share in the soil biomass, has been estimated as small (e.g., Andren and Schnurer, 1985; De Ruiter *et al.,* 1993; Didden *et al.,* 1994), but indirect contributions, affecting the functioning of other groups of organisms or even the structure and functioning of the food web as a whole, have been postulated to be important (Moore *et al.,* 1988; Bouwman *et al.,* 1994; Brussaard *et al.,* 1995; Beare, 1997).

Soil fauna contribute to litter breakdown by (i) grinding plant residues and (ii) channeling and improving the soil structure (Shaw and Pawluk, 1986; Cheshire and Griffiths, 1989; Scheu and Wolters, 1991; Coûteaux *et al.,* 1995). Tian *et al.* (1995a) found that for plant materials with contrasting chemical composition in the field, an increased breakdown rate occurred following the addition of earthworms or millipedes. These workers suggested that the role of soil fauna was relatively greater in the decomposition of materials with a high C/N ratio, lignin, and polyphenol content and less important on low C/N ratio residues as these residues were decomposed easily by microorganisms. Termites and ants are also known to be efficient in digesting cellulose-containing substances and, in some cases, lignified substances (Lee and Wood, 1971; Wood, 1988). Millipedes break down plant litter and mix it with mineral soil, which they ingest (Kevan, 1968; Tian *et al.,* 1995a).

House *et al.* (1984) and Stinner *et al.* (1988) proposed that with conservation tillage, surface-maintained crop residue can provide a continuous resource in space and time for many decomposer organisms. Friebe and Henke (1991) reported that a higher tillage intensity was associated with a lower faunal abundance, whereas a distinct increase in plant–residue decomposition was associated with a reduced tillage intensity.

In another study, Reddy *et al.* (1994) found that the residue mass loss was significantly correlated to moisture content and arthropod abundance. However, there is little information on the combined effects of abiotic and biotic factors on the mass loss of plant material during the process of decomposition.

The microbial decomposer community is extremely diverse and capable of surviving a wide range of environmental and food-related stresses. Bacteria and fungi, the major decomposers, differ in their mode of growth and activity (Coleman and Crossley, 1996). Bacteria are clustered in colonies occupying a small soil volume and their movement in soil is episodic and related to factors such as rainfall, root growth, tillage, and ingestion by soil fauna (Gregorich and Janzen, 1998). In contrast, fungi have hyphae that can grow over relatively larger distances and penetrate into small spaces where they can decompose OM by secreting enzymes and translocate the nutrients back through hyphae.

The species of microorganisms responsible for residue breakdown also depends on the temperature of the substrate (Parr and Papendick, 1978) and the type of substrate (Wani and Shinde, 1977; Wardle and Lavelle, 1997). Actinomycetes are responsible for residue breakdown mainly at high temperatures, whereas species of bacteria and fungi are dominant at lower temperatures (Parr and Papendick, 1978). Wani and Shinde (1977) observed that fungi were responsible for the favorable effect on the mineralization of C during the decomposition process, with *Aspergillus* species playing a more important role in the breakdown of cellulose. Several studies (Hendrix *et al.,* 1986; Holland and Coleman, 1987; Neely *et al.,* 1991; Wagner and Broder, 1993; Baldy *et al.,* 1995) have reported that fungi tended to be the dominant microorganism controlling litter decomposition.

Ingham and Horton (1987) showed that for high C/N ratio soils and materials, the fungal/bacterial ratio increased markedly while the decomposition proceeded, the bacteria presumed to have peaked early in the process, probably had a diminishing role after the first week. Thus, when the more easily decomposable constituents of the residue have been exhausted, bacteria may continue at a lower level, perhaps feeding on fungal carbohydrate products (Wagner and Broder, 1993), and the decomposition was predator controlled (Vreeken-Buijs and Brussaard, 1996).

6. Management Factors

a. Loading Rate and Quantity Added

Evidence from laboratory and field studies has generally demonstrated that the rates of decomposition of plant materials added to soil are proportional to amounts added initially (Pinck and Allison, 1951; Jenkinson, 1965, 1977; Larson *et al.*, 1972). According to Pinck and Allison (1951), the percentage decomposition of plant C in soils was nearly independent of input levels, provided that C additions did not exceed the amount equivalent to 1.5% of the dry soil. In addition, Ladd *et al.* (1983) found that the proportions of plant ^{14}C and ^{15}N *Medicago littoralis* retained in soil were not independent of the amounts of plant ^{14}C and ^{15}N added. These workers concluded that the greater the amount of labeled plant material added, the smaller the proportion of residual organic ^{14}C and ^{15}N.

Some workers reported that small amounts of crop residues decompose more rapidly in soil than large amounts. This was supported in some, but not all, experiments with ^{14}C-labeled plant materials (Jenkinson, 1971). Sorensen (1963), however, concluded that the percentage decomposition of organic materials was nearly always independent of the quantity added if the C addition did not exceed 1.5–2.0%. Summerell and Burgess (1989) also reported no difference in the rate of straw decomposition at two loading rates of 1.6 and 3.2 tons ha^{-1}.

Stott *et al.* (1990) reported that out of three loading rates of 1680, 3360, and 6720 kg ha^{-1} of surface residues added to a soil, the lowest loading rate showed the fastest loss rate. Likewise, Broadbent and Bartholomew (1948) reported the inverse relation between decomposition and quantity of residue added. However, these workers could not find any reason for slow decomposition at a higher loading rate. Perhaps these differences may be related to N availability in soil. If, in certain conditions, N is limiting, it may be sufficient for hastening the decomposition of a small amount of low-quality residues but may slow down the decomposition if large amounts of residues are added.

b. Method, Mode, and Accessibility

The accessibility of plant residues to soil microbes is of primary importance in its rate of decomposition. The method of application of plant residues, such as residue particle size and placement, can provide a different degree of accessibility, which, in turn, affects residue breakdown rates as well as the mineralization–immobilization process.

Many studies indicate that burying residues in soils increases the decomposition rate compared to placing residues on the soil surface (Douglas *et al.*, 1980; Harper and Lynch, 1981; Summerell and Burgess, 1989; Aulakh *et al.*, 1991a; Douglas and Rickman, 1992; Buchanan and King, 1993; Schomberg *et al.*, 1994). The effect of placement decreases with time (Christensen, 1986; Cogle *et al.*, 1987). For example, Cogle *et al.* (1989) observed that incorporated straw decomposed faster than surface straw during the first 15 days but the differences abated thereafter. The initial lag in the decomposition of residues on the soil surface is probably due to litter on the surface being more subjected to unfavorable conditions for decomposition, particularly with respect to fluctuations in temperature-and moisture-limiting microbial activity than in material buried in soil. The incorporation of straw allowed microorganisms decomposing the residues greater access to nutrients in the soil and factors favoring their activity (Harper and Lynch, 1981). Also rainfall leaches soluble nutrients out of the straw, reducing the microbial activity further. Douglas *et al.* (1980) reported that the rate of decomposition for surface residues was much more linear with time than buried residues and that surface residues tended to decompose entirely, such as phase II of buried residues as proposed by Berg and Agren (1984).

Schomberg *et al.* (1994) compared the decomposition and N dynamics of surface placed and buried lucerne, wheat, and sorghum residues under different moisture regimes. Their results showed that large differences in residue decomposition occurred between crop residues, with lucerne residue decomposing much faster than wheat and grain sorghum with net N mineralization occurring throughout the study period. Net N immobilization was longer than 1 year for surface wheat and sorghum residues, but was about 0.33 a year when buried. Both N_{max} (g N immobilized kg^{-1} of original biomass) and N_{eq} (g N immobilized kg^{-1} of biomass loss) were influenced by crop and placement. The N_{max} value was similar for surface wheat and sorghum residues, but was 50% lower for buried wheat than for sorghum. N_{eq} indicated that the N requirement of microorganisms was less for buried than for surface residues. Results showing a rapid decomposition of incorporated residues compared to surface residues have also been reported by other workers (e.g., Lafond *et al.,* 1996; Beare, 1997).

Kanal (1995) found that increasing the depth of residue incorporation from 50 to 200 mm resulted in a decrease in the breakdown rate due to less biological activity. In contrast, Breland (1994) found that increasing the incorporation depth up to 300 mm increased the decomposition rate due to a more favorable moisture regime in lower layers.

Residue left over from burning generally decomposed slowly because the C remaining after burn is less active biologically and has a longer turnover time (Rasmussen and Parton, 1994) and is partly due to the decrease in the decomposer population after burning (Prasad and Power, 1991). A lower microbial biomass with burning (Collins *et al.,* 1992) tends to support this contention. Shindo (1991) reported that charred residues evolved much less CO_2 than noncharred residues when incubated with soil.

c. Irrigation

Irrigation in drier regions assists the rapid decomposition of plant residues. Douglas *et al.* (1980) found that 65% of the buried wheat straw was decomposed under dryland conditions in eastern Oregon in 14 months compared with 70–80% in 13 months reported by Smith and Douglas (1971) in Idaho under irrigated conditions.

d. Crop Cover

Numerous decomposition studies (mostly laboratory incubation and some field experiments) have been performed without the presence of plants in the soil. The presence of plants in the soil system interacts with the decomposition and soil N processes by modifying the soil environment physically, chemically, and biologically (Clarholm, 1985). A plant cover reduced straw weight loss rates (Christensen, 1985b), which agreed with field results using ^{14}C-labeled plant materials (Jenkinson, 1977; Shields and Paul, 1973). The plant effect was attributed to the drier conditions in planted soils due to plant water uptake, but alternative explanations have been offered (Reid and Goss, 1982). Christensen (1985b) found that both summer soil temperatures and soil moisture were reduced in planted plots. Growing plants may reduce moisture contents by water uptake and by intercepting part of the precipitation. During the relatively drought summer conditions, both these processes may cause planted soils to be drier than fallowed soils. The plant effect on soil temperature arises from their higher reflectance, interception of incoming solar radiation, and cooling provided by water transpiration. The reduced moisture and temperature levels of planted soils may reduce microbial activity, extractable N and N mineralization, and slow residue decomposition (Faber and Verhoef, 1991; Parmelee *et al.,* 1993).

The possibility that roots exert some control over residue decomposition has been investigated for many years (Gadgil and Gadgil, 1971; Dighton, 1991; Zhu and Ehrenfeld, 1996). Results showed that in soils with low total C and N contents, roots stimulated greater activity of the soil biota (Parmelee *et al.,* 1993, 1995), which contributed, in turn to faster litter decomposition and nutrient release (Zhu and Ehrenfeld, 1996).

In other studies involving crop plants in agricultural soils, plant residue decomposition in soil was enhanced by living roots or, in turn, the presence of a crop (Dighton *et al.,* 1987; Sallih and Bottner, 1988; Cheng and Coleman, 1990; Dormaar, 1990). This may be due to root exudates inducing increased microbial activity, although Whitley and Pettit (1994) found that the addition of lignite humic acid was responsible for increases in decomposition.

Roots, rhizodeposits, and exudates have attracted little decomposition research effort, especially compared to plant inputs to soil from aboveground parts. This is despite the importance of plant–soil interface and their input in C flow; 16–33% of the total C assimilated by plants is released directly into soil by roots, which contributes to 30–60% of the organic C pool in soil (Boone, 1994). This has particular importance in arable agriculture where root residues may be the only organic inputs and major contributors to SOM replenishment. The advent of C isotope methods has significantly improved the knowledge of decay rates and allowed discrimination between root and microbial respiration (Cheng *et al.,* 1994; Swinnen *et al.,* 1995). In general, the decomposition of root products conforms with the general pattern of aboveground plant parts, being determined by the chemical composition (Andren *et al.*, 1990; Swinnen *et al.*, 1995).

7. Climate Factors

Temperature and rainfall are important factors affecting the decomposition of OM and plant residues (Schomberg *et al.*, 1996). The loss of OM in warm regions where OM decomposes rapidly is of a serious concern. Using ^{14}C-labeled ryegrass, the rate of decomposition was found to be about four times faster in Nigeria (humid tropics) as compared to England (humid temperate) (Jenkinson and Ayanaba, 1977; Barnard and Kristoferson, 1985).

Temperature and moisture regimes have been found to be good predictors of the microbial C/total C ratio (Insam *et al.*, 1989). A low temperature in winter and a dry soil in summer limit microbial decomposition, whereas microbial decomposition is greatest during the moist warm spring and autumn seasons (Douglas *et al.*, 1990; Collins *et al.*, 1990b).

At a regional level, the effects of climate on decomposition are reflected in the accumulation of OM in the soil. Data on SOM from semiarid, semihumid, and humid regions of the world show that soils from warm climates contain less OM than that in cool regions, partly because of faster decomposition (Jenny, 1941).

8. Other Factors

a. Elevated Ozone and Carbon Dioxide

Plant residues grown under elevated ozone (O_3) and/or CO_2 have been shown to affect plant structure and nutrient status and, as a result, their decomposition (Boerner and Rebbeck, 1995; Cotrufo *et al.*, 1994). Elevated atmospheric CO_2 could affect decomposition in one of four ways (Ball, 1997) through

- a direct effect on microbial activity
- changes in plant species composition of an ecosystem
- changes in the quantity of plant residues produced
- changes in the chemical composition of residues

There is more likely to be a general decline in litter quality than other effects in response to increased CO_2 (Korner and Miglietta, 1994; Arp *et al.*, 1997; Torbert *et al.*, 1998), as enhanced atmospheric CO_2 is widely known to increase photosynthesis, but with an uncertain compensatory uptake of nutrients. This results in a reduction in nutrient concentration in plant tissues (Heal *et al.*, 1997; Arp *et al.*, 1997). Henning *et al.* (1996) found that although elevated CO_2 increased the C/N ratio and lignin content of sorghum stem and soybean leaves, it had no impact on soil C turnover, relative N mineralization, cumulative C and N mineralization, and C/N mineralized, thus suggesting that increasing atmospheric CO_2 will have little effect on composition or decomposition of field crop residues. Because CO_2 enrichment results in increased photosynthetic C fixation, the possibility exists for increased soil C storage under field crops in an elevated CO_2 world (Henning *et al.*, 1996).

For details regarding the effects on processes and magnitude of decomposition as affected by elevated CO_2 levels, readers are advised to read the review by Ball (1997). Briefly, Ball concluded that the growth of C3 plants in elevated atmospheric CO_2 (600–700 μmol mol^{-1}) may lead to a significant increase in either or both C/N and lignin/N ratios of plant residues (Table II). Short-term decomposition of litter from plants showing this response in elevated CO_2 has confirmed that decomposition occurs at a significantly lower rate (Table II). However, growth responses and degradability of C4 plants suggested no difference in litter quality or degradability (Table II). Further long-term studies are needed to confirm these effects in complex ecosystems; if these results are repeated at the ecosystem level, then significant changes in the cycling of C and N may occur as a result of elevated CO_2.

b. Ultraviolet (UV) Radiations

The impact of UV radiations on litter quality and decomposition presents an analogous situation to CO_2 elevation as increasing levels of UV radiation at higher latitudes were found to cause changes in plant biochemical composition, with increased pigmentation and secondary compounds (Heal *et al.*, 1997). Increased tannins and decreased α-cellulose in the leaf of *Vaccinium* spp. have been recorded in experimentally increased UV radiation studies (Gehrke *et al.*, 1995; Johnson *et al.*, 1995). There was evidence of a small direct effect of UV in reducing decomposition and changes in fungal community structure. Because direct UV exposure affects only superficial litter, the main effects are anticipated to be through an alteration in litter quality (Heal *et al.*, 1997). However, the combination of direct and indirect effects could cause a reduction in the decomposition rate of the order of 5–10% at higher latitudes.

B. METHODS OF STUDYING CROP RESIDUE DECOMPOSITION

The range and objectives within studies of crop residue decomposition have generated a wide variety of methods used to study residue decomposition, each appropriate for a specific purpose (Harper, 1989; Conteh *et al.*, 1997). Some of the commonly used methods are reviewed here.

1. Measurement of Carbon Dioxide Evolution

In this method, the loss of C (as CO_2) is measured following the decomposition of soil and crop residues incubated together minus the loss of C from soil in-

Table II

Nutrient Ratios of Residues and Soil Respiratory Activity in Soils Amended with Plant Litter from C3 and C4 Plants Grown in Elevated CO$_2$[a]

Species	Ambient CO$_2$	Elevated CO$_2$
	C/N ratio of plant residues	
C3 plants		
Acer pseudoplatanus L.	81	106
Betula pubescens	35	53
Fraxinus excelsior	42	56
Lolium perenne	17	30
Picea sitchensis	18	18
Triticum aestivum	42	79
C4 plants		
Andropogen gerardii	102	96
Poa pratensis	33	39
Sorghastrum nutans	104	99
Sorghum bicolor	14	14
Spartina patens	81	77
	Lignin/N ratio of plant residues	
C3 plants		
A. pseudoplatanus L.	16	22
B. pubescens	10	17
F. excelsior	5	9
L. perenne	16	25
P. sitchensis	5	6
T. aestivum	5	6
C4 plants		
A. gerardii	38	36
P. pratensis	10	11
S. nutans	33	31
S. bicolor	6	6
S. patens	28	28
	Soil respiration (mg Co$_2$-C g soil^{-1} day^{-1}) with crop residues	
Soil only (control)	75	75
C3 plants		
Scirpus olneyi	183	147^b
Triticu aestivum	180	129^b
Lolium perenne	216	162^b
C4 plants		
S. patens	165	150
S. bicolor	156	171

[a] Colated from reviewed data by Ball (1997).

[b] Significantly different at $P < 0.05$.

cubated in the absence of added residues (Stevenson, 1986; Scheu, 1993; Bremner *et al.*, 1991; Hassink, 1994b; Ladd *et al.*, 1995). These laboratory incubations were conducted under favorable temperature and moisture regimes and often excluded processes such as faunal activity and nutrient leaching, which influence the decomposition in the field. Nevertheless, the techniques used are useful for characterizing the influence of individual factors on decomposition such as inorganic matrices (Skene *et al.*, 1997), sodicity and salinity (Nelson *et al.*, 1996), elevated CO_2 (Gorissen *et al.*, 1995), residue composition (Vanlauwe *et al.*, 1996), N addition (Green *et al.*, 1995), and the presence of living roots (Cheng and Coleman, 1990; Nicolardot *et al.*, 1995). This approach, however, has limitations, including analytical errors in measuring the long-term release of CO_2, particularly when the amount of plant residues added is kept small, relative to the amount of native SOM. Also, the assumption is made that the addition of plant residues to the soil does not alter the decomposition rate of the native SOM (Stevenson, 1986).

In the CO_2 evolution method, evolved CO_2 from microbial respiration is absorbed in an alkali solution, precipitated with $BaCl_2$ and subsequently estimated by titrating against standard HCl, assuming that 1 ml of 1 *M* HCl releases 22 mg CO_2 (Ladd *et al.*, 1995). Reactions of the process are as follows:

$$2OH^- + CO_2 \longrightarrow CO_3^- + H_2O$$

$$Ba_2{}^+ + CO_3^- \longrightarrow BaCO_3 \text{ (solid)}$$

The rate of CO_2 production is used to describe the rate of plant residue decomposition (Hassink, 1994b; Ladd *et al.*, 1995; Saggar *et al.*, 1996).

Nordgren (1988) developed an apparatus for the continuous, long-term monitoring of the soil respiration rate in a large number of samples. The method is based on the principle that a decrease in electrical conductivity is directly proportional to the amount of CO_2 absorbed in the alkali solution. The method gives accurate measurements and is rapid. The amount of CO_2 absorbed in the alkali solution is calculated as in Eq. (4):

$$C_t = a\,(1 - R_0/R_t). \tag{4}$$

C_t denotes absorbed CO_2 at time t and a is the proportionality constant relating decrease in conductance to absorbed CO_2. The solution electrical conductance at time t is R_t and R_0 at $t = 0$.

Apart from the automatic operation, the conductivity method has several advantages per se. The system is closed and involves no tubing and air flows and each recorded observation is an average of about 1000 readings.

Many variations exist in the methods used for measuring CO_2 (Alef and Nannipieri, 1995) such as the use of biometer flasks [Organization for Economic Cooperation and Development (OECD), 1981; Haigh, 1993) and, more recently, infrared (IR) gas analyzers (Alef and Nannipieri, 1995).

These CO_2 evolution methods have also been used to characterize the decomposition in field conditions using both static chambers and dynamic chamber methods (Jensen *et al.*, 1996). The advent of portable IR gas analyzers made it possible to measure the rate of CO_2 evolution within a short time in the field (Alef and Nannipieri, 1995; Magid *et al.*, 1997b). Jensen *et al.* (1996) compared the static chamber method using alkali trapping of CO_2 for 24 hr and a dynamic method using IR gas analysis for 2 min at each point for estimating field scale CO_2 fluxes from unplanted soils. They concluded that the static method provided the best integrative measure, but it would be necessary to measure continuously in order to obtain a better estimate of CO_2 evolution using the dynamic method. However, a major limitation of the static chamber method is that it does not accurately reflect high rates of CO_2 evolution from the soil due to a diffusional limitation of the trapping of CO_2 into the alkali. Thus the initial CO_2 evolution from the decomposition of freshly added plant material will often be considerably underestimated by this method (Magid *et al.*, 1997b).

2. Recording Weight Loss Using Mesh Bags

The mesh bag method was developed to elucidate decomposition in undisturbed soil systems and, because of its simplicity, was extended to arable systems, in which plant residues are normally admixed with the soil by tillage practices (Christensen, 1985a).

Mesh bags containing straw or a soil–straw mixture have been used most frequently to study plant residue decomposition (Brown and Dickey, 1970; Smith and Douglas, 1971; Harper and Lynch, 1981; Douglas and Rickman, 1992; Thomas and Asakawa, 1993; Schomberg *et al.*, 1994; Cotrufo *et al.*, 1995; Cortez *et al.*, 1996; Jama and Nair, 1996; Vreeken-Buijs and Brussaard, 1996). The total weight loss of straw enclosed in mesh bags is attributed to three loss components: leaching, microbial decomposition, and loss of straw particles through mesh openings. The initial weight loss is probably determined by leaching and microbial decomposition of a readily available substrate. In the following stages, decomposition proceeds and straw becomes more fragile, eventually disintegrating and causing a loss of straw particles from the mesh bags.

Mesh bags can be criticized for providing conditions potentially dissimilar from that in bulk soil and have a number of limitations (Heal *et al.*, 1997): only net changes are measured and the fate of material leaving the bag is ignored, thus overestimating the decomposition. Also, this material may be more labile and contain more nutrients than the residues remaining and may play an important role in nutrient cycling processes. In order to minimize these difficulties, small mesh sizes are normally used, but apart from the analytical procedures connected with the contamination of the plant residues by soil, the confinement or isolation from soil can generate a microenvironment that differs considerably from that of residues in a more intimate contact with soil (Malkomes, 1980). Residues within the litterbags have been reported to have more moisture compared to unbagged residues and are thus more favorable for microbial activity (Vossbrinck *et al.,* 1979). Also, when bags are used in arable soils, straw does not suffer the physical damage likely to result from contact with agricultural machinery during incorporation (Harper, 1989). Tian *et al.* (1992a) found a positive correlation between the decomposition rate and the mesh size of litterbags that was interpreted as an effect of increasing accessibility to soil fauna. Fine meshes exclude soil animals that may influence decomposition (Jensen, 1985; Vreeken-Buijs and Brussaard, 1996), although House *et al.* (1987) reported that the decomposition rate was independent of mesh size.

Another problem arises from the significant quantities of soil adhering to the decomposing residues. Various cleanup procedures, such as brushing and washing with water (Brown and Dickey, 1970; Douglas *et al.,* 1980; Harper and Lynch, 1981; Schomberg *et al.,* 1994; Cotrufo *et al.,* 1995; Vreeken-Buijs and Brussaard, 1996) have been used, but methods are not satisfactory as they most likely provide selective and incomplete removal of adhering soil components (Brown and Dickey, 1970). Moreover, washing procedures undoubtedly remove water-soluble OM and nutrients from the residues (Douglas *et al.,* 1980; Christensen, 1985a).

In some studies using mesh bags, the recovery of straw samples has been corrected for its ash content, with the weight loss being expressed on an ash-free dry weight basis (Harper and Lynch, 1981; Thomas and Asakawa, 1993; Schomberg *et al.,* 1994; Jama and Nair, 1996; Vreeken-Buijs and Brussaard, 1996). This approach does not account for SOM introduced into the straw sample by soil entering the litterbags.

In most cases, mesh bags probably underestimate the actual breakdown rates. The exclusion of certain macrofauna components is a justified criticism when the absolute rate of weight loss and N immobilization–mineralization is required. The continuing popularity of the mesh bag method results primarily from the relative ease with which samples of known initial weight can be recovered and used to study nutrient dynamics under field conditions.

3. *In Vitro* Profusion Methods

This method uses a closed, continuous air flow system in which the plant material is allowed to decompose in a medium of constant ionic strength (Nyami, 1992). The evolved CO_2 is absorbed in an alkali solution and reactions involved are similar to those discussed earlier.

The rate of CO_2 production is used to describe the rate of plant residue decomposition. However, the profusion apparatus of Nyami (1992), which was made from glass, is expensive to make and difficult to use. Using simple components used in health care, Lefroy *et al.* (1995) developed an apparatus at the University of New England, Australia, that provides comparable results and is simple to use.

Although this technique has not been compared with actual decomposition data obtained from the field, it allows the screening of a wide range of crop residues under a standardized set of conditions, without the complications of differences in soil biota, and allows a ranking of relative breakdown and nutrient release.

4. Lignin Analysis

Because lignin decays relatively very slowly (Scheu, 1993; Rutigliano *et al.*, 1996), the weight loss of decomposing residues may be estimated from the increase in lignin content. Although initial weight loss could be estimated by lignin content, there was increasing deviation from unity as decay proceeded, probably resulting from lignin decay at later stages (Harper, 1989). Also, lignin decomposition can be enhanced by earthworms (Scheu, 1993), which may lead to the underestimation of residue decomposition. Thus, these methods need calibration relating the periodic increase or decrease in lignin content with decomposition under a wide variety of soils, environments, and other biotic factors (Harper, 1989).

5. Size Density Fractionation for *in Situ* Measurements of Decomposition

Magid *et al.* (1996) proposed that decomposition can be estimated by determining the temporal distribution of plant residues in different density and size-density fractions of OM. This approach was used successfully to study the decomposition of rape straw, and estimates of decomposition from particulate organic matter (POM) were in qualitative agreement with those based on field-scale CO_2 fluxes (Magid *et al.*, 1997a). This method shows considerable potential but needs to be tested for different kinds of residues. It can be applied without much problem to residues that are predominantly insoluble in water, as recovery of the POM fraction necessitates a decantation or flotation step (Magid *et al.*, 1997b). However, the technique leads to a mixing of added residues with native SOM, thus requiring an unamended control treatment. A major advantage of this technique is that added residues are completely exposed to the soil environment and thus the full range of faunal and other soil interactions.

6. Isotopic Techniques

A method of studying residual decomposition in the field is to add labeled (^{15}N, ^{14}C, ^{13}C, etc.) plant residues to the soil. Labeled residues are placed in small cylinders to prevent contamination by unlabeled materials. At periodic intervals, soil is removed from the cylinder and analyzed to measure the quantity of label remaining. This approach permits the direct measurement of the fate of plant litter, but is usually limited to small-scale studies because of the cost of establishment and analysis.

a. Nitrogen-15 Techniques

Introduction of the ^{15}N technique has provided a means for reexamining earlier concepts. The ^{15}N technique has been used in both laboratory and field experiments in the following types of studies:

- Gross rates of N mineralization associated with the decomposition of plant residues (Watkins and Barraclough, 1996).
- As an indicator of SOM turnover (Ladd, 1981; Tiessen *et al.,* 1984; Muller and Sundman, 1988; Cadisch *et al.,* 1993; Green and Blackmer, 1995).
- Using ^{15}N nuclear magnetic resonance (NMR) in decomposition studies (Knicker and Lüdemann, 1995).

b. Carbon-14-Labeled Residues

^{14}C-labeled organic compounds have been used extensively to study the rate of decomposition and stability of microbial products (Voroney *et al.,* 1991; Amato and Ladd, 1992; Alvarez *et al.,* 1995; Cheshire and Chapman, 1996; Hassink and Dalenberg, 1996; Nelson *et al.,* 1996; Saggar *et al.,* 1996). The use of ^{14}C-labeled residues makes it possible to follow the decomposition of added residues with considerable accuracy, even in the presence of large amounts of native SOM. It has also been possible to identify plant C as it becomes incorporated into fractions of the soil humus (Stevenson, 1986).

$^{14}CO_2$ evolved from ^{14}C labeled crop residues is absorbed in alkali solution as described earlier, and the radioactivity of absorbed $^{14}CO_2$ is determined with a scintillation counter (Amato *et al.,* 1984; Ladd *et al.,* 1995; Saggar *et al.,* 1996).

^{13}C NMR (Inbar *et al.,* 1989; Baldock and Preston, 1995; Preston, 1996) and bomb ^{14}C techniques have also been used in studying the annual inputs of C, the rate of residue decomposition, and turnover time (O'Brien and Stout, 1978; O'Brien, 1984; Goh, 1991).

c. Variation in Natural Stable Carbon Isotope Ratio ($^{13}C/^{12}C$)

The stable C isotope ratio ($^{13}C/^{12}C$) of SOM is very similar to that of the vegetation growing on it. Plants with a C3 photosynthesis pathway have a low $^{13}C/^{12}C$ ratio compared to those with C4 pathway, and thus at sites where the vegetation has changed from one photosynthetic pathway type to the other, the $^{13}C/^{12}C$ ratio of SOM changes and this change can be measured (Gregorich *et al.,* 1996). This approach has been used to evaluate the decomposition in field studies (Balesdent and Mariotti, 1996), but is limited to sites where information on the time of change of vegetation is known. However, the method has considerable potential as it characterizes C produced *in situ,* which can be used to evaluate soil C dynamics over short (1 year) or long (>100 years) time periods (Gregorich *et al.*, 1996; Gregorich and Janzen, 1998).

d. Sulfur-35 and Phosphorus-32

The short half-lives of ^{32}P (14 days), ^{33}P (25 days), and ^{35}S (87 days) make them less appropriate than ^{14}C (half life = 5730 years) for studies of SOM dynamics, particularly in long-term studies in the field (Lefroy *et al.,* 1995). Despite these limitations, these techniques have been used successfully in studies of the breakdown of the residues labeled with ^{33}P (Friesen and Blair, 1988), ^{31}P NMR (Gressel and McColl, 1997) and ^{35}S (Lefroy *et al.,* 1994) and in the dynamics of ^{35}S in fractions of SOM (Eriksen *et al.,* 1995).

7. Other Methods

The strength of decomposing wheat internode walls as measured by needle penetrometers (Harper, 1989), breaking strength of residue (McCalla, 1943; Armburst, 1980), water, and alkali soluble color of decomposing residues (Pauli, 1970; Harper and Lynch, 1981; Harper, 1989) has been determined and related to the decomposition of residues, but these methods generally work well only during initial stages of decomposition.

C. Modeling Decomposition of Crop Residues

Theoretical and predictive models have been developed to describe the decomposition process mathematically. Most of these models were based on the concepts of microbial activities. These concepts have been abstracted and simplified by Paustian *et al.* (1997) as shown in Table III. The simple first-order model has been used most frequently to characterize the decomposition of crop residues (Meentemeyer, 1978; Aber and Mellilo, 1982; Table III), where the decomposition rate was related to substrate remaining and thus mass loss followed the exponential decline. Single component first-order decay models [Eq. (5)] was used by Jenny (1941) to describe N loss in cultivated soil as

$$dX/dt = A - k\,X, \tag{5}$$

where X is the soil organic C or N content, A is the addition rate, and k is the first-order rate constant (i.e., the fraction of soil C or N decomposed each year). This model was fitted to residue decomposition data from several field experiments and the calculated turnover times (1/k) varied from 18 to 36 years (Gregorich and Janzen, 1998). The use of first-order kinetics to describe decomposition implied

Table III

Overview of Some Litter Decomposition Models and Their Representation of Litter Quality Effects on Decomposition[a]

Model type/author	No. of litter compartments	Elements	Decomposer biomass	Litter quality representations/other comments
Single pool-models				
Aber and Mellilo (1982)	1	C, N	No	Lignin to N
Berg and Ekbohm (1991)	1	C	No	Finite asymptote included
Janssen (1984)	1	C	No	Decomposability defined by "apparent initial age"
Meentemeyer (1978)	1	C	No	Lignin; used for regional comparisons
Middleburg (1989)	1	C	No	Decomposability defined by "apparent initial age," specific rate constant decreases with time
Multiple litter pool models				
Berg and Agren (1984)	2	C	No	Flux from resistant to labile component to represent solubilization
Jenkinson (1977)	2	C	No	Two-pool determination from curve fitting
Minderman (1968)	6	C	No	Approximate analyses to determine initial pool size
Moorhead and Reynolds (1991)	4	C, N	Yes	Labile, holocellulose, and resistant initial litter fractions; includes lag time due to microbial colonization
Parnas (1975)	2	C, N	Growth rate only	C and N and C only compounds distinguished
Andren and Paustian (1987)	4	C	No	Soluble and insoluble fractions used for initial pool sizes; formation of secondary products
Continuous spectrum models				
Carpenter (1981)	na[b]	C	No	Quality parameter derived from curve fittting
Bossatta and Agren (1991, 1994)	na	C, N, P, S	Steady state	Quality parameters have been related to chemical fractionations
Boudreau (1992)	na	C	No	Gamma distributions for reactivities

[a] From Paustain *et al.* (1997).

[b] Not applicable.

that the metabolic potential of the soil microbial biomass exceeds the substrate supply; more explicitly, changes in microbial biomass are not associated with changes in the rate of decomposition (Gregorich and Janzen, 1998). This model treated the litter as a uniform, homogeneous substrate and explained only short-term decomposition rates (1 year) (Paustian *et al.,* 1997) as it is well known that different plant components are not uniform in quality and different litter constituents decompose at different rates (Berg and Agren, 1984; Moorhead and Reynolds, 1991). This was also shown in radio-carbon dating studies showing that the age of SOM ranged from hundreds to thousands of years (Campbell, 1978; Stout *et al.,* 1981; Goh *et al.,* 1984). To explain different decomposition rates, multiple litter pool models have been developed based on the inherent decomposability and the factors that influence decomposition rates of different constituents of litter (Table III). Such multiple models generally assumed that each litter constituent had its own potential decomposition rate and each component decomposed independently according to first-order relation (Table III; Berg and Agren, 1984; Moorhead and Reynolds, 1991). In these models the most common litter quality attributes were defined chemically (e.g., lignin, N content, C/N ratio, cellulose, polyphenols).

Juma and McGill (1986) defined the litter components as structural and metabolic components, having slow versus rapid decomposition rates, respectively. In an alternative approach, litter was considered as a continuous distribution of organic materials rather than a set of discrete pools (continuous spectrum models; Table III). This approach incorporated both the influence of initial litter composition and the transformation of primary litter compounds into secondary materials and subsequent effects on the overall decomposition rate (Carpenter, 1981; Bossatta and Agren, 1991, 1994).

In another novel variation of the multiple first-order pool models, Sinsabaugh and Moorhead (1994) proposed that decomposition rates were directly related to enzyme activity and they measured litter quality by assays of potential enzyme activity.

An important conclusion from these models was the realization that in order to predict total mass loss and the decomposition of primary litter fractions, the formation and turnover of secondary decay products need to be included. This led to the development of OM turnover models, where OM pools representing secondary decomposition products (e.g., microbial biomass, SOM) are included (e.g., CENTURY, Parton *et al.,* 1987; Rothamstead model, Jenkinson *et al.,* 1987; CERES model, Godwin and Jones, 1991). The SOM and N submodels were used where litter OM and N were partitioned into different compartments according to decomposabiltity based on chemical composition. These submodels, which predict decomposition and the flow of inorganic and organic nutrients, are combined with soil water and plant growth submodels to simulate cropping system dynamics.

IV. CROP RESIDUES AND MANAGEMENT PRACTICES

Several major options available to farmers in the management of crop residues include (i) burning, (ii) incorporation, (iii) direct drilling in surface residues, (iv) undersowing crops, and (v) baling and removing crop residues.

i. Residue Burning. One of the main advantages of burning crop residues is that it clears the land quickly of residues before the next crop is established, especially in high-intensity cropping areas, thus facilitating seed germination and establishment. The extent to which residues are burnt influences its effectiveness, but this varies depending on the method, timing of burning, and moisture content of residues. Total cover burning is preferred for controlling residue-borne diseases (Staniforth, 1982; Butterworth, 1985) but this is seldom achieved.

When cool, wet conditions prevail after harvest, a good burn will not be achieved if the residues are too wet. Felton *et al.* (1987) advised that when burning of straw was inevitable, it should be burned out after seasonal rain periods to minimize leaching losses of nutrients released after burning.

ii. Incorporation of Residues. Crop residues may be incorporated partially or completely into the soil depending on methods of cultivation used (Dormaar and Carefoot, 1996). This results in various degrees of ground cover by the residues, which has important implications for controlling water and wind erosion. In certain parts of the United States where erosion is prevalent, farmers are required to carry out "conservation tillage" to maintain a minimum ground cover by residues of 30% (Griffith and Wollenhaupt, 1994).

Ploughing is the most efficient residue incorporation method (Ball and Robertson, 1990; Christian and Bacon, 1991). Methods of incorporation can affect yield and decomposition. Deep incorporation may reduce possible yield depressions, but slows down cultivation and requires more time, labor, and energy costs (Chaman and Cope, 1994).

iii. Direct Drilling in Surface Mulched and Unmulched Residues. Direct drilling is a practice that leaves straw residues from a previous crop on the soil surface without any form of incorporation; the residues may or may not be mulched. The following crop is then drilled directly into the soil with residues left on the soil surface. The large volume of residues remaining on the surface often leads to machinery failure, thus affecting the proper sowing of the seeds of the following crops (Staniforth, 1982; McGuigan, 1989).

iv. Undersowing Crops. This involves either undersowing a cereal crop with clover or drilling grass, forage legumes, or green manures into cereal stubble (Scott *et al.,* 1973; Badaruddin and Meyer, 1989; Deo *et al.,* 1993). Grazing animals are also important in this system as they usually graze off and or trample the cereal stubble after its harvest (Fraser and Francis, 1996) and contribute to the nutrients cycling through their dung and urine returns (Haynes and Williams, 1993).

v. BALING AND REMOVING STRAW. Concurrent with the movement toward reduced tillage systems in production agriculture is the increasing need for value-added processing from outside agriculture as a means of diversifying production and stabilizing income levels (Stumborg *et al.,* 1996; Latham, 1997; Powell and Unger, 1997). Surplus straw from agriculture (especially poor quality) may be used for a number of useful purposes such as stock feed, fuel, building material, livestock bedding, composting for mushroom cultivation, bedding for strawberries, cucumbers, melons, and other crops, mulching for orchards, and sources of chemicals. For example, the Cochrane group (1994) estimated that the net return to the producer in the United States was \$20.84 less the cost of nutrients lost in the straw (\$4.81 ton^{-1}) based on values of crop residues and their acquisition costs. These estimates did not include the value of crop residues in increasing or maintaining SOM and soil structure.

A. EFFECTS OF RESIDUES AND MANAGEMENT ON SOIL QUALITY

1. Soil Quality Indicators

The increasing awareness of the progressive degradation of soils has led to the search for a reliable measure of soil quality. Traditionally, soil productivity has been used as a measure for soil quality (Hornick, 1992). More recently, the concept of soil quality has been suggested by several authors as a tool for assessing the long-term sustainability of agricultural practices at local, regional, national, and international levels (Lal, 1991; Sanders, 1992; Papendick and Parr, 1992; Parr *et al.,* 1992; Karlen *et al.,* 1992; Acton and Padbury, 1993; Doran and Parkin, 1994; Gregorich *et al.,* 1994). Soil quality has been defined as "the capacity of a soil to take up, store, and recycle water, minerals, and energy so that crop production is maximized and environmental degradation is minimized" (Trasar-Cepeda *et al.,* 1998).

Brookes (1989) stated that a high-quality soil should (i) assist in the reduction of contaminant levels in surface and subsurface waters, (ii) allow the production of healthy and nutritious crops, and (iii) display key characteristics of ecosystem maturity. Soil quality depends on a large number of physical, chemical, biological, and biochemical soil properties, and its characterization requires the selection of the properties most sensitive to the management practices as soil quality indicators (Elliott, 1994; Doran *et al.,* 1994; Cameron *et al.,* 1996).

Crop residue management is known to either directly or indirectly affect most of these indicators. It is perceived that soil quality is improved by the adoption of crop residue management practices. Karlen *et al.* (1994) evaluated several proposed soil quality indicators and developed a "soil quality index" based on several parameters. Their results gave ratings of 0.45, 0.68, or 0.86 for removal, normal, or double residue treatments, respectively. Effects of crop residue management on soil properties are reviewed herein.

2. Soil Physical Properties

Crop residues play an important role in maintaining good soil physical conditions. In most climates, the removal of all crop residues from the field leads to a deterioration of soil physical properties (Kladivko, 1994).

a. Soil Erosion

The presence of crop residues on the soil surface is known to reduce both wind and water erosion of soil either directly by affecting the physical force involved in erosion or indirectly by modifying the soil structure through the addition of organic matter (Brown *et al.*, 1989; Franzluebbers *et al.*, 1996; Lafond *et al.*, 1996). Crop residue amendments have been shown to restore yields on desurfaced or artificially eroded soils (Dormaar *et al.*, 1988; Larney and Janzen, 1996), probably due to an increase in soil aggregate stability (Cresswell *et al.*, 1991; Sun *et al.*, 1995).

Flat residues as a mulch on the soil surface act as a barrier restricting soil particle emission from the soil surface and also increasing the threshold wind speeds for detaching these particles. It has been reported that standing residues are more effective than flat residues in reducing erosion by reducing the soil surface friction velocity of wind and intercepting the saltating soil particles (Hagen, 1996). The greater the amount of residues left on the surface, the greater the reduction in wind erosion (Michels *et al.*, 1995).

Many studies reported that incorporating residues to various degrees can reduce runoff and hence water erosion losses of soil by 27–90% (Freebairn and Boughton, 1985; McGregor *et al.*, 1990; Cassel *et al.*, 1995; Fawcett, 1995). However, these reductions mainly occur where considerable amounts of residues remain on the soil surface after incomplete incorporation (Freebairn and Boughton, 1985; McGregor *et al.*, 1990; Dormaar and Carefoot, 1996).

b. Soil Aggregation and Soil Structure

Information on the effect of crop residue management on soil aggregation is limited, although it is well established that OM helps maintain aggregate stability (Tisdall and Oades, 1982; Oades, 1984; Hamblin, 1987; Boyle *et al.*, 1989; Haynes *et al.*, 1991; Haynes and Francis, 1993). The addition of crop residues is expected to have a positive effect on soil structure and aggregation (Freebairn and Gupta, 1990).

Nuttall *et al.* (1986) reported over a period of 6 years in the Canadian prairies that aggregates >0.84 and <12.7 mm in size were most abundant with crop residues chopped and spread and were least abundant with autumn (fall) ploughing. Values for spring burning of crop residues were intermediate. However, other residue management treatments (removal, incorporation, twice the amount incorporated and burning) did not significantly affect aggregation in a 14-year study at Kansas (Skidmore *et al.,* 1986). However, Karlen *et al.* (1994) found that the normal rate of crop residues increased soil aggregation following no-till corn compared to removal and that the doubling of crop residue amounts increased soil aggregation and stability significantly.

Singh *et al.* (1994) reported greater amounts of large water-stable aggregates in no-till + straw treatments. The proportion of wind-erodible (<1 mm) and water-slakable microaggregates (<0.25 mm) was also lower, and mean weight diameter (MWD) and geometric mean diameter (GMD) were greater compared to tillage treatments.

An increase in GMD of aggregates has been observed within a week when small amounts of residue were added due to a flush of fungal growth, but when a large amount of residue was added, the increase in GMD was observed after only 6 weeks (Hadas *et al.,* 1994). It is suggested that size and strength of aggregates apparently caused by fungi increased during the first week due to external reinforcement by hyphae, whereas changes appearing after only 6 weeks were attributed to bacteria and due to internal reinforcement by bacterial secretions (Hadas *et al.,* 1994). Significantly higher levels of ergosterol, a sterol related to fungal biomass (Eash *et al.,* 1994), were found in plots receiving crop residues. This suggests that long-term crop residue treatments were affecting fungal populations at this site and that the quality of residues also affected the formation and stability of aggregates (Hadas *et al.,* 1994). Thus the proper management of residues can provide farmers with some measure to mitigate changes due to implements and traffic loads imposed during the cropping cycle (Tate, 1987) and also improves soil structure and aggregation.

c. Compaction, Bulk Density, and Penetration Resistance

In the mixed pasture-arable cropping system, there are more chances of soil compaction due to farm equipment traffic (Wagger and Denton, 1989; Unger, 1986) and annual traffic (Abel-Magid *et al.,* 1987; Dao *et al.,* 1994). Under the weight of soil mass, the impact of raindrops, or the compactive pressure of traffic, soil particles reorient themselves and pack together more tightly, while excluding air and water contained within, resulting in higher bulk density and penetration resistance. However, surface and subsurface soil density and penetration resistance may increase naturally when using a no-tillage system (Ehlers *et al.,* 1983; Mielke *et al.,* 1986) resulting from the raindrops effect and the structural failure (collapse) of soils having low-stability aggregates (Bautista *et al.,* 1996). Soils with a high sand content are especially prone to develop a dense zone with high penetration resistance (Awadhwal and Smith, 1990).

The effect of residue management and tillage has been found to be variable. Some workers reported no effect (Blevins *et al.,* 1977; Hill, 1990; Ismail *et al.,* 1995) whereas others found lower soil bulk densities in a conservation tillage-residue management system (Edwards *et al.,* 1992), residue incorporation (Sidhu and Sur, 1993), and no-tillage surface residue (Dao, 1996). Bulk density has been observed to be higher in conservation tillage and no-tilling residue fields, as soils continually consolidated in the absence of tillage or with shallow tillage (Voorhees and Lindstrom, 1983; Pikul and Allmaras, 1986; Larney and Kladivko, 1989; Rasmussen and Smiley, 1989; Unger, 1986).

In most studies, increases in soil bulk density and compaction were reported in crops that were seeded in comparatively wide rows (0.7 to 1.0 m), such as corn (Johnson *et al.,* 1989; Vyn and Raimbault, 1993), soybean (Fahad *et al.,* 1987), or sorghum (Bruce *et al.,* 1990) crops.

d. Soil Hydraulic Conductivity and Infiltration

Crop residues increase soil hydraulic conductivity and infiltration by modifying mainly soil structure, proportion of macropores, and aggregate stability. These increases have been reported in treatments where crop residues were retained on the soil surface or incorporated by conservation tillage (Murphy *et al.,* 1993). Up to eightfold increases in hydraulic conductivity in zero-tillage stubble retained have been reported over treatments where stubble was removed by burning (Bissett and O'Leary, 1996; Valzano *et al.,* 1997). Hydraulic conductivity under straw-retained direct drilled treatments was 4.1 times greater than that of straw-burnt conventional tillage treatments (Chan and Heenan, 1993).

Observations (Hanks and Anderson, 1957; McMurphy and Anderson, 1965) of frequently burned rangelands in Kansas Flint Hills in the United States indicated that annual burning reduced the infiltration rate. These differences could be related to the effect of fine noncapillary porosity of the topsoil, decreases in porosity of crust (Veckert *et al.,* 1978, Pikul and Zuzel, 1994), or the development of water-repellent soil surface from hydrophobic compounds of plant residue (Scifres and Hamilton, 1993). Another reason may be leaching of fine particle of ash or dispersed clay to lower layers and clogging the pores, breaking the continuity of pores (Araujo *et al.,* 1994). This decrease in infiltration rate results in greater runoff and erosion (Robichand and Waldrop, 1994).

Baumhardt and Lascano (1996) observed increases in cumulative infiltration from 29 mm for bare soil to as high as 49 mm under different residue and management practices. Cassel *et al.* (1995) reported that tillage practices that leave crop residues on the soil surface can reduce or eliminate surface crusting, increase infiltration, and reduce surface runoff and soil loss while increasing crop yields.

Thus, although increases in soil hydraulic conductivity and infiltration can result through proper residue management practices, care should be taken because increased infiltration may lead to leaching, causing nutrient losses and groundwater pollution.

e. Soil Temperature

Crop residue management influences soil temperature significantly. Major mechanisms involved are (a) a change in radiant energy balance and (b) insulation (Unger and McCalla, 1980). The radiation balance is influenced by the heating of air and soil, the evaporation of soil water, and the effect of incoming radiation by surface residues (Van Doren and Allmaras, 1978). Residue characteristics involved in the reflectance of incoming radiations include residue age, color, orientation, distribution, and amount (Unger and McCalla, 1980). The insulation effect of crop residues is controlled by the amount and associated thickness of residue cover.

In arid and semiarid regions or in summers, crop residues left on the soil surface as a mulch as compared to incorporation, removal, or burning are known to be beneficial for crop production (Dao, 1993; Tian *et al.,* 1993b). This reduces the soil temperatures, thus influencing the biological processes (Hatfield and Prueger, 1996), and enhances soil N mineralization (Tian *et al.,* 1993a). In temperate areas, residue burning, removal, or incorporation provided greater yields because residues left on the soil surface reduced the seed zone soil temperatures, resulting in poor or delayed germination and poor crop growth and grain yields (Schneider and Gupta, 1985; Kaspar *et al.,* 1990; Burgess *et al.,* 1996; Swanson and Wilhelm, 1996). The removal of residue from a wider strip of row area may benefit plant growth in some areas by increasing soil temperature in the row (Kaspar *et al.,* 1990; Swan *et al.,* 1996).

f. Soil Moisture Content

It has been well established that increasing amounts of crop residues on the soil surface reduce the evaporation rate (Bussiere and Cellier, 1994; Gill and Jalota, 1996; Prihar *et al.,* 1996). Thus, residue-covered soils tend to have a greater moisture content than bare soils except after extended drought (Tanaka, 1985; Thomas *et al.,* 1990; Felton *et al.,* 1995; Cantero-Martinez *et al.,* 1995; Bissett and O'Leary, 1996; Moitra *et al.,* 1996; Peterson *et al.,* 1996). Residue mulch or partial incorporation in soil by conservation tillage has also been shown to increase the infiltration by reducing surface sealing and decreasing runoff velocity (Box *et al.,* 1996). Studies have shown that soils retained more moisture when residues were retained on the soil surface by conservation tillage as compared to residue incorporation, removal, or burning (Osuji, 1984; Freebairn *et al.,* 1986; Unger, 1986; Dormaar and Carefoot, 1996). Boyer and Miller (1994) reported a 27 and 18% decrease in surface and subsurface soil water-holding capacities, respectively, in burned treatments as compared to nonburn. The amount of residue cover is also important in determining the moisture retention in soil (Power *et al.*, 1986a; Wilhelm *et al.*, 1986).

In some studies, however, workers failed to obtain a significant difference in soil moisture content between no-tillage residue retained and conventional tillage with residue incorporated, removal, or burning (Hill *et al.*, 1985; Nuttall *et al.*, 1986). These results may occur because of extended dry periods or because amounts of residue present were too low to be significant.

3. Soil Chemical Properties

a. Soil pH

One of the most important factors determining soil fertility is pH, which may, however, be influenced strongly by cultivation and crop residue management. It has been noted in Australia, New Zealand, and many other parts of the world that soil pH decreases as a result of continuous cultivation of clovers and other leguminous crops (Mengel and Steffens, 1982; Juo *et al.*, 1996) attributed mainly to BNF (Bolan *et al.*, 1991) and to proton release by legume roots (Schubert *et al.*, 1990), resulting in the accumulation of organic anions such as malate, citrate, and oxalate in plants (Bolan *et al.*, 1991).

Research has shown that if these organic anions are returned to the soil and on decomposition by microorganisms, soil pH can be increased due to the decarboxylation of organic anions (Yan *et al.*, 1996), ligand exchange (Hue and Amien, 1989), and addition of basic cations (Bessho and Bell, 1992). Thus, one possible way of protecting soil from acidification is by returning the crop residues to the soil (Miyazawa *et al.*, 1993; Yan *et al.*, 1996). Kretzschmar *et al.* (1991) showed that the field application of crop residues for 6 years increased soil pH significantly from 4.54 to 5.69. This was also reported by other workers (e.g., Hue and Amien, 1989; Bessho and Bell, 1992; Hafner *et al.*, 1993; Karlen *et al.*, 1994). Increases in pH by pearl millet straw (*Pennisetum americanum*) have been reported to decrease aluminium toxicity in acidic soils (Kretzschmar *et al.*, 1991). Increases in soil pH occurred irrespective of whether crop residues were burnt, incorporated, or mulched (Kretzschmar *et al.*, 1991; Ball-Coelho *et al.*, 1993; Kitou and Yoshida, 1994). In some areas, tillage-induced reductions in pH have been reported, but the direct drilling through surface residue showed only a minor decrease in soil pH (Smettem *et al.*, 1992).

Differences in the magnitude of pH change may be because different plants species differ in their capacities in accumulating organic anions. Legumes accumulate higher amounts of organic anions than grasses (Mengel and Steffens, 1982). Legume residues often induced greater increases in soil pH than grasses or other crop residues (Hue and Amien, 1989; Bessho and Bell, 1992; Miyazawa *et al.*, 1993). However, plants contain a large amount of organic N, such as proteins and amino acids, which can be mineralized to nitrate in soils producing protons during nitrification and hence acidifying the soils (Yan *et al.,* 1996).

Increases in pH after burning were generally attributed to ash accretion (Raison, 1979; Goh and Phillips, 1991; Bauhus *et al.,* 1993; Araujo *et al.,* 1994; Van Reuler and Janssen, 1996) as ash residues are generally dominated by carbonates of alkali and alkaline earth metals but also contain variable amounts of silica, heavy metals, sesquioxides, phosphates, and small amounts of organic and inorganic N (Raison, 1979).

b. Soil Organic Matter and Nitrogen

Accumulation of SOM is a reversible process and most current agricultural practices are responsible for its reduction in agroecosystems (Jenkinson, 1981; Rasmussen and Collins, 1991; Heenan *et al.,* 1995) with the consequent decrease of soil biological fertility and soil resilience (Lal, 1994) through an impoverishment of physical, chemical, and biological properties of soils (Kirchner *et al.,* 1993; Wood and Edwards, 1992; Perucci *et al.,* 1997). This can only be practically compensated by burying the crop residues in the soil. Thus, the practice of residue incorporation may be better able to sustain arable soils and represents an interesting method of managing soil fertility (MacRae and Mehuys, 1985; Prasad and Power, 1991; Geiger *et al.,* 1992; Aggarwal *et al.,* 1997). This has implications on amounts of soil microbial biomass as well as labile soil C and N content (McGill *et al.,* 1986; Ross, 1987; Gupta *et al.,* 1994; Heenan *et al.,* 1995; Campbell *et al.,* 1996a,b), thereby influencing the turnover of soil organic matter and consequently the availability of nutrients for crops (Aggarwal *et al.,* 1997; Perucci *et al.,* 1997).

The addition of crop residues on OM and N increases may depend on the amount of residue added, quality of the residue environment, and the duration of addition. Short-term addition in hot climates, promoting rapid decomposition, may lead to only a slight or no increase in soil OM (Aggarwal *et al.,* 1997), but long-term addition has been shown to increase both C and N contents (Karlen *et al.,* 1994). The effect of crop residues on SOM content is related strongly to the amount of residues added and only weakly related to the type of residue applied (Rasmussen and Collins, 1991). Earlier studies in the United States (Larson *et al.,* 1972), Canada (Sowden, 1968), and Germany (Sauerbeck,1982) concluded that different types of crop residues had similar effects on SOM and that it is more a function of microbial product recalcitrance than initial residue composition (Voroney *et al.,* 1989).

Several studies showed that organic C and N in soil responded linearly to an increased rate of residue addition (Larson *et al.,* 1972; Black, 1973; Rasmussen *et al.,* 1980; Karlen *et al.,* 1994). Soil C and N were found to decrease with time for all residue additions except manure (Fig. 1), and the rate of decrease was related to the level but not the type of residue returned to the soil (Fig. 2) (Rasmussen and Collins, 1991). Losses of OM will continue in many of the present

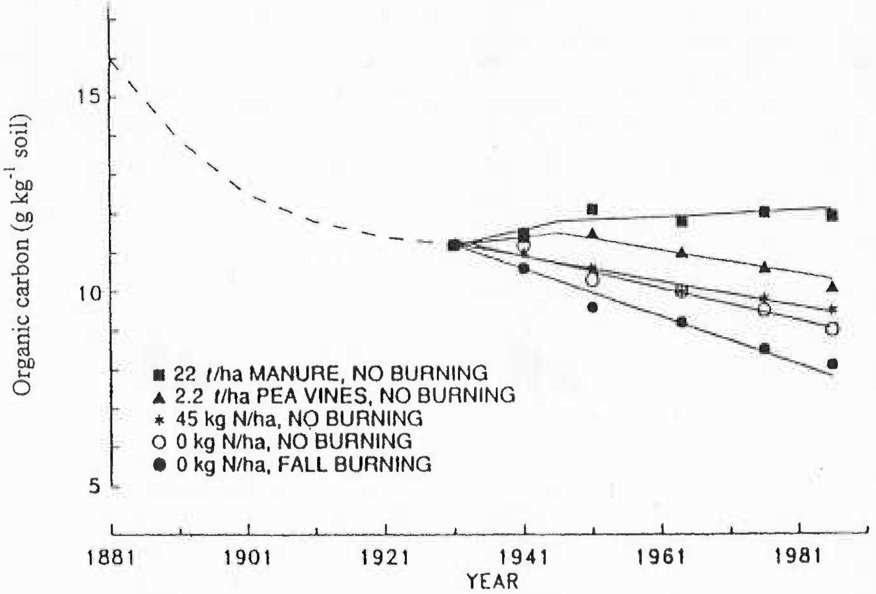


Figure 1 Effect of management practices on long-term changes in organic C in the top 30 cm of a haploxeroll soil in Oregon. From Rasmussen and Collins (1991).

cropping systems without adequate amounts of residue return to soil. While residue input may increase OM content, continued input must be sustained (Rasmussen and Collins, 1991) as some long-term studies have shown that OM and N increased compared to removal or burning (Karlen *et al.,* 1994; Dalal *et al.,* 1995; Perucci *et al.,* 1997).

Many reviews comparing different management strategies such as no-tillage, conservation tillage, and conventional tillage on conserving soil C and N have been reported (Prasad and Power, 1991; Rasmussen and Collins, 1991). In general, conservation tillage increased the amount of soil C and N as reported in many parts of the world (Table IV). More recently, greater C and N have been reported under no-tillage or conservation tillage compared to mould board plough or conventional tillage (Campbell *et al.,* 1995, 1996a,b). These increases were greater in clay soils compared to sandy soils (Campbell *et al.,* 1996a,b) due to the physical protection of organic matter by clay (Van Veen and Paul, 1981; Hassink and Whitmore, 1995). In other studies, increases in C and N were reported but were confined to the soil surface (25 mm) (Dalal *et al.,* 1991, 1995; Angers *et al.,* 1997). However, Angers *et al.* (1997) found that C and N contents in 0- to 10-cm soil layer were higher under no-tillage compared to ploughing, whereas in deeper layers (20–40 cm depth), the trend was reversed. When all soil depths were combined, no differences between treatments were found in humid soils of eastern Canada.

Likewise, no effects between tillage treatments were reported by Franzluebbers and Arshad (1996) from Canada and by Fettell and Gill (1995) from red brown earth soils in Australia.

The small or even no effects reported in some studies may be due to the high initial C and N content in soils, making it difficult to detect significant increases in C and N as a result of treatments against a large background (Campbell *et al.*, 1991a,b,c; Rasmussen and Collins, 1991). For example, using a site under subterranean clover pasture for some years with high initial C and N levels, Heenan *et al.* (1995) showed that under subterranean clover–wheat rotation, direct drill residue-retained treatment increased soil C and N content continuously, whereas under either grazing or cultivation, C and N were reduced. In all other treatments, C and N decreased. These decreases were more when residues were burnt or when soil was cultivated. The effects were additive in burnt and tilled soils. Whether in lupin–wheat or continuous wheat, residue retention and direct drilling showed a smaller decrease in the C and N ratio compared to cultivation and burning (Heenan *et al.*, 1995).

Decreases in C and N contents have been observed under residue burnt treatments (Pikul and Allmaras, 1986; Biederbeck *et al.*, 1980; Wood, 1985; Collins *et al.*, 1992; Heenan *et al.*, 1995). Shultz (1992) showed that burning caused a decline of C and N by 6% overall for wheat/legume rotation and continuous cereal and wheat fallow rotation, whereas a 1% increase was obtained under stubble retention. However, in other studies, no effect of burning compared to residue incorporation on soil C content was reported (Nuttall *et al.*, 1986; Rasmussen *et al.*, 1980). However, soil N levels were reduced by residue burning (Rasmussen *et al.*, 1980). This may be due to incomplete burning as up to 33% of the C in charred

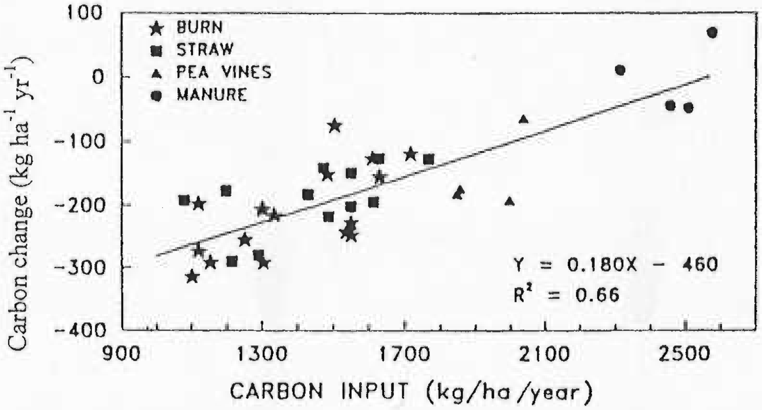


Figure 2 Effect of the rate of carbon input on organic C change in a haploxeroll soil in Oregon. From Rasmussen and Collins (1991).

Table IV

Effect of Conservation Tillage on Organic C and N in Soil in Different Parts of the World[a]

Location and soil	Annual precipitation (mm)	Soil depth (cm)	Duration of study (year)	Tillage system[b]	Increase (% year^{-1})	
					C	N
South Africa						
Haploxeralf	412	10	10	TT	5.6	3.4
Haploxeralf	412	10	10	NT	7.3	5.1
Germany						
"Podsol"	—	30	5	NT	3.2	1.4
"Podsol"	—	30	5	NT	2.4	1.6
"Podsol"	—	30	6	NT	1.3	1.3
Australia						
Western						
Psamment	345	15	9	NT	1.6	—
Alfisol	307	15	9	NT	0.7	—
Alfisol	389	15	9	NT	1.4	—
Queensland						
Pellustert	698	10	6	NT	1.2	1.3
Canada						
Saskatchewan						
Chernozem	—	15	6	NT	6.7	2.8
United States						
North Dakota	375	45	25	SM	1.8	1.3
Haploboroll	375	45	25	SM	−0.1	0.1
Haploboroll						
Argiboroll	375	45	25	SM	0.5	0.4
Kansas	—	15	11	NT	0.7	0.6
Haplustoll						
Nebraska	446	9	15	NT	2.8	2.4
Haplustoll	446	10	15	NT	1.2	1.0
Oregon	416	15	44	SM	0.3	0.4
Haploxeroll						
Washington	560	5	10	NT	1.9	2.0
Haplxeroll						
Mean	—	—	—	—	2.2	1.7
Minimum	—	—	—	—	−0.1	0.1
Maximum	—	—	—	—	7.3	5.1

[a] From Rasmussen and Collins (1991).

[b] TT, tine till; NT, no till; SM, stubble mulch.

residues remained on the soil surface after burning and the mulch of the charred residues left was not biologically active (Rasmussen and Collins, 1991). It is likely that burning changes the quality rather than the quantity of OM in soil.

c. Phosphorus

Most workers reported increased phosphorus (P) accumulation near the soil surface in no-tillage or minimum tillage systems (Langdale *et al.,* 1984; Follett and Peterson, 1988; Weil *et al.,* 1988). Stratification of P resulting from minimum tillage is believed to result in improved P availability because there is less soil contact of organic P in crop residues and hence less P fixation (Blevins *et al.,* 1983). Burning cereal residues also resulted in a higher extractable P content in the surface 0- to 2.5-cm soil layer (Nuttall *et al.,* 1986) and increased plant uptake (Van Reuler and Janssen, 1996). Compared to residue removal, the incorporation of residues of cluster bean, mung bean, and pearl millet has been found to increase available P, probably due to an increase in phosphatase (both acid and alkaline) enzyme activity (Aggarwal *et al.,* 1997).

d. Other Nutrients

Most studies on plant decomposition have focused on N dynamics, with little or no emphasis on other nutrients that are important in describing the crop response to decomposing organic residues on soils with marginal fertility. Even if residue decomposes quickly, nutrients contained in it are not subjected to the same rapid loss as that which occurred under burning (Luna-Orea *et al.,* 1996). Instead, nutrients are released over time by chemical, physical, and biological processes. Considerable quantities are released within a short period of time depending on the residue quality (Luna-Orea *et al.,* 1996) and the kind of nutrients (Lefroy *et al.,* 1995). Increases in exchangeable cations (K, Ca, Mg) and base saturation have been reported (Kretzschmar *et al.,* 1991; Geiger *et al.,* 1992).

The greater availability of both macro- and micronutrients has been reported more under conservation tillage than other conventional tillage (Hargrove *et al.,* 1982; Langdale *et al.,* 1984; Hargrove, 1985; Follett and Peterson, 1988; Edwards *et al.,* 1992). The burning of crop residue has also resulted in an increased K content in the surface soil compared with no burning, crop residue removal, or incorporation (Moss and Cotterill, 1985). Although burning induces short-term increases in nutrients, losses of nutrients due to burning can occur.

e. Loss of Nutrients Due to Burning of Crop Residues

The burning of crop residues can result in nutrient loss as a result of the direct convective transfer of ash (Harwood and Jackson, 1975), and subsequent losses may be increased by the action of wind and water. Simultaneous losses of C, S, and N have been reported on residue burning (Marschner *et al.,* 1995).

However, very little is known about the effects of burning intensity and frequency on nutrient losses and dynamics. O'Connor (1974) indicated that in a fairly dense tall tussock grassland stand containing 175 kg N ha^{-1} above ground, about 40 kg N ha^{-1} was lost when such grasslands were burnt every 3 years. In some European studies, an 8 kg N ha^{-1} year^{-1} loss was reported on burning postharvest cereal stubble and a loss of 10 kg N ha^{-1} $year^{-1}$ occurred when grass or legume pastures in west Africa were burnt (Winteringham, 1984). A loss of 2600, 4800 kg C ha^{-1} and 742 kg N ha^{-1} has been reported during pre- and postharvest burning of sugarcane trash, respectively (Ball-Coehlo *et al.,* 1993). Animal grazing of pastures has been shown to conserve N loss due to burning as N losses from burning grazed pastures (9 kg ha^{-1}) were half of those (18 kg N ha^{-1}) from the burning of ungrazed pastures (Hobbs *et al.,* 1991). Likewise, S losses from high S- and low S-containing rice crop residues were 60 and 40%, respectively (Lefroy *et al.,* 1994).

High N losses in the form of ammonia volatilization can also occur in soils after stubble burning because of alkaline ash left on the soil surface and an increase in urease activity (Bacon and Freney, 1989). Emissions of ammonia from straw burning were calculated to be equivalent to 20 ktons N $year^{-1}$ in the United Kingdom in 1981, which declined to 3.3 ktons $year^{-1}$ in 1991 as a result of changes in agricultural practices because of an imposed ban on burning crop residues (Lee and Atkins, 1994). The fraction of total plant N released as ammonia was estimated to be between 40 and 80%. These data suggest that emissions of ammonia from straw and stubble burning constituted a significant but declining source of ammonia that is currently not accounted for in European and North American emission inventories.

If combustion of plant residues is nearly complete, most of the C, H, O, N, and organic S and P are transferred to air, whereas many of the cationic elements are rendered water soluble and hence are readily available to plants (Raison, 1979). Where combustion is inhibited for any reason, the ash will be black and contain residual OM (Jordan, 1965).

In general, losses of nutrients due to burning decrease in the order of N > Ca > S > K > Mg > P > Na. These losses depend on the temperatures reached during burning. Volatile losses of P and K occur at temperatures exceeding 500°C, whereas the vaporization temperature of Na is reported to be 880°C (Raison, 1979). However, these temperatures may not be achieved during the burning of cereal crop residues and grass/legume pastures, and most of these elements are left in the ash. In forestry systems, where large amounts of wood and residues are burnt, higher burning temperatures were reported and greater losses of nutrients occurred (Feller, 1988; Gillon and Rap, 1989; Goh and Phillips, 1991). Also, nutrients that were left in the ash were highly soluble in water and may be prone to leaching and runoff losses (Mangas *et al.,* 1992; Malmer, 1996).

4. Soil Biological Properties

Leaving crop residues on the soil surface or incorporation provides a favorable environment for soil and surface residue dwelling organisms because of reduced water loss, amelioration of temperature extremes, fluctuations, and the presence of a relatively continuous substrate for decomposers (House and All, 1981; Crossley *et al.,* 1984; Wardle and Lavelle, 1997). Several aspects of crop residue management practices on soil biological properties have been studied.

a. Soil Biota

i. MICROBIAL BIOMASS/MICROORGANISMS. The soil microbial biomass (SMB) is the living component of the soil that comprises mainly fungi and bacteria, including soil microfauna and algae. Although it accounts for only 1–3% of organic C and 2–6% of organic N in soil (Jenkinson, 1987), it plays a key role in SOM and nutrient dynamics by acting as both a sink (during immobilization) and a source (mineralization) of plant nutrients. Because SMB and microbial activity are closely related to SOM content, they are influenced positively by organic amendments such as crop residues and animal manures (Ocio *et al.,* 1991; Collins *et al.,* 1992; Nannipieri *et al.,* 1994). Perucci *et al.* (1997) reported that SMB-C content was significantly greater when crop residues were incorporated in the soil instead of removed. The changes in SMB-C were also related to the type of residues incorporated (Perucci *et al.,* 1984). Biederbeck *et al.* (1980) reported a decline in microbial biomass when residues were removed by burning. In an similar study, burning of residues reduced SMB to 57% of that in the manured treatment, probably because of the smaller amount of OM returned to the soil (Collins *et al.,* 1992).

In addition to crop residues, the no-till system also affects SMB by promoting C accumulation at the soil surface due to a lack of incorporation of crop residues. Results from many studies found that SMB was significantly higher in the surface of no-till than in conventional tilled soils (Doran, 1980; Linn and Doran, 1984; Buchanan and King, 1992; Angers *et al.,* 1993a,b). However, taking into account the whole plow layer, smaller differences existed as shown by Franzluebbers *et al.* (1995), where a 75 to 146% greater SMB-C concentration under no-till than conventional tillage at a 0- to 50-mm depth existed but only 12 to 43% greater SMB-C concentration at the 0- to 200-mm depth was present in several wheat experiments.

Higher populations of bacteria, *Actinomycetes,* fungi, earthworms, and nematodes have been reported in residue mulch than in incorporated residues (McCalla, 1958). Residue incorporation produced more microbial activity than residue removal or burning (Beare *et al.,* 1996). Ten to 80% greater aerobic microorganisms and 60 to 300% greater anaerobic bacteria (including denitrifiers) were reported in surface soils under reduced than conventional cultivated soils (Doran, 1980). This is expected because soil beneath residue has a better microenvironment as compared to conventional tillage. Gupta and Germida (1988) obtained similar results and found fungi to be particularly sensitive to tillage effects. Direct counts of viable bacteria and total fungal hyphae were consistently higher on residue from incorporated treatments compared with removed or burned treatments (Beare *et al.,*1996).

Burning of residue has been shown to favor bacteria rather than fungi (Miller *et al.*, 1955; Jones and Richards, 1978). Biederbeck *et al.* (1980) found that long-term burning decreased the microbial population of the soil permanently. At one location, a decline of 85% in the fungal and 70% in the bacterial population was reported. Raison (1979) also reported a severe decline in the bacteria population involved in nitrification due to burning. The extent of reduction in microbial populations would depend on the soil temperature rise during burning. Except in fires, where large amounts of accumulated fuel were burnt, heat production is intense but for a short duration. In addition, because only 5% of heat energy released may be partitioned to the soil (Packham, 1969), maximum temperature may not occur near the soil surface but at a variable height in the vegetation (Kenworthy, 1963; Goh and Phillips, 1991). Temperatures below the soil surface cannot exceed 100°C until all moisture evaporates (Aston and Gill, 1976). Goh and Phillips (1991) reported a temperature of 60°C at the soil–humus interface and, in a few cases, it was as high as 110°C. In the soil, increases in temperature are usually less than 50–80°C and are restricted to the top 3–4 cm of soil, which persisted for only a few minutes (Lawrence, 1966), and does not always reduce the microbial populations except in forestry systems where large quantities of fuel are burnt.

Effects of ash on soil biological properties have not been studied extensively. Goh and Phillips (1991) attributed the increase in ammonification and nitrification on burning to increase, in humus pH and increases in the population of ammonifying organisms, possibly due to the elimination of other competing organisms for the same substrate. In another study, ashed soils have been found to provide conditions favorable to nitrifying bacteria (Bauhus *et al.*, 1993). Decreases in microbial biomass and soil respiration have been reported from treatments where residues were burned or removed in comparison to straw-returned treatments (Gupta *et al.*, 1994; Karlen *et al.*, 1994; Ladd *et al.*, 1994; Beare *et al.*, 1996).

In addition to the residue management practice, residue quality has been found to affect the microbial population as less bacterial and fungal populations were reported on grass/cereal residues as compared to legumes. The difference shown is substrate-induced respiration (Parmelee *et al.*, 1989; Beare *et al.*, 1996). Robinson *et al.* (1994) observed that leaves and internodes of wheat (differing in C/N ratio and physical structure) support vastly different fungal communities. This is true especially for fungi, which depend on resource quality (Wardle *et al.*, 1993), but bacteria seem to be less affected compared to fungi (Cornejo *et al.*, 1994; Wardle, 1995). Biological assays have shown that bacterial communities were also affected as total substrate activity and functional diversity were higher for bacteria in incorporation treatments compared with burning treatments (Beare *et al.*,1996).

ii. Soil Fauna. Many studies have shown that earthworms and microarthropods may assume a more dominant role in OM decomposition and nutrient flux patterns in residue management systems (Prasad and Power, 1991; Tian *et al.*, 1993b; Vreeken-Buijs and Brussaard, 1996). Predatory and saprophagus soil arthropods, as well as crop-damaging herbivores such as ground beetles (*Coleoptera carabidae*), spiders, and decomposer fauna such as earthworms, are reported to be in large numbers in no-tillage residue management treatments compared to conventional tillage treatments (Edwards, 1975; Barnes and Ellis, 1979; House and All, 1981; Blumberg and Crossley, 1983; House and Parmelee, 1985; Tian *et al.*, 1993b; Chan and Heenan, 1995; Buckerfield and Webster, 1996). This increase in macroorganism population and activity under residue retention is due to the fact that the mulch is known to attenuate the increase of soil temperature, decreased diurnal variations, and to retain higher soil moisture in addition to providing food for soil animals (Tian *et al.*, 1993b; Hartley *et al.*, 1994; Buckerfield and Webster, 1996). The benefits of an organic mulch on soil surface were demonstrated by significantly higher moisture (34% increase), whereas earthworm density increased by 155% over bare soil (Buckerfield and Webster, 1996). Results from many studies suggest that straw retention increased the population of earthworms and other macroorganisms compared to straw removal or burning (Nuutinen, 1992; Karlen *et al.*, 1994).

The burning of crop residues has been shown to reduce both the activity and the population of macroorganisms and it takes several months to 5 years to recover that activity (Neumann and Tolhurst, 1991; Tisdall, 1992; DeCaens *et al.*, 1994; Doube *et al., 1994*). It has been shown that direct drilled plots had greater population density and biomass of earthworms and cocoons than tilled plots. On direct drilled plots, burning stubble resulted in smaller adult earthworms and a lower density of cocoons (Doube *et al.*, 1994), whereas plots with standing stubble had fewer and smaller adults than where stubble was in close contact with soil.

Although crop residue management and their effect on microclimate play a role in determining the population of macroorganisms, the nutritional quality of plant material appears to be more important in influencing earthworm population (Tian *et al.*, 1993b). As plant residues with different chemical composition vary in their palatability for soil fauna, they are expected to have different effects on soil faunal populations (Tian *et al.*, 1992b). In examining the food preference of leaf litter by earthworm, Hendriksen (1990) found that the earthworm population was significantly and negatively correlated with the C/N ratio and polyphenol concentration of plant materials. Similar results have been reported by Tian *et al.* (1992b), where earthworm populations were negatively correlated to the lignin/N ratio of plant residues, and the population of ants was also significantly correlated to the N concentration of plant residues (Tian *et al.*, 1993b).

iii. MYCORRHIZA. Mycorrhiza play an important role in plant growth and crop production (Bethlenfalvay, 1992). However, effects of soil management practices on the abundance and activity of mycorrhiza are complex. It is reported that tillage disrupts soil networks of mycorrhizal hyphae and may impair nutrient uptake and reduce crop yields (Evans and Miller, 1990). Cropping with hosts that become colonized extensively with mycorrhiza may affect the quality of crops produced (in terms of relatively greater nutrient uptake) and improve the quality of the crop residues produced. This may improve the soil quality indirectly by improving the aggregate stabiltiy, microbial biomass, and other parameters. However, there is little published evidence on the effects of tillage and residue management on mycorrhizal associations with plants and their effects on soil and plant quality need to be explored.

b. Microbial Processes

i. ENZYMES. Crop residues and tillage have been reported to significantly and rapidly (within few weeks) alter the composition, distribution, and activity of the soil microbial community and enzymes (Doran, 1980; Dick, 1984; Magan and Lynch, 1986; Martens *et al.,* 1992; Scagnozzi *et al.,* 1995; Deng and Tabatabai, 1997). Although straw amendments also contain enzymes, the increase in activity in soils with organic residues most likely results from the stimulation of microbial activity rather than the direct addition of enzymes from the organic sources.

Soil enzyme activities also respond to tillage practices. Gupta and Germida (1988) compared soils cultivated for 69 years with adjacent grasslands and found that cultivation depressed phosphatase activity by 49% and arylsulfatase activity by 65%. Significantly greater activities of acid phosphatase, alkaline phosphatase, arylsulfatase, invertase, amidase, and urease in surface soils under no-till plots have been observed (Dick, 1984). Also, activities of amidohydrolases, glycosidases, and arylsulfatase were generally greater in soil under no-till or mulch treatments compared to chisel plough and mould board plough with or without crop residues (Deng and Tabatabai, 1996a,b, 1997). A higher activity of both acid and alkaline phosphatase and dehydrogenase enzymes due to residue incorporation compared to residue removal or burning treatments has also been reported (Aggarwal *et al.,* 1997).

Differences in enzyme activities may be due to differences in the origin, states, and/or persistence of different groups of enzymes in soils (Deng and Tabatabai, 1997). Alkaline phosphatase in soils is believed to be derived entirely from microorganisms because it has not been found in plants (Juma and Tabatabai, 1977; Tabatabai, 1994). Soil pH also controls either the rate of enzyme activity or the stability of some enzymes (Tabatabai, 1994). Significant correlations found between the activity of different enzymes (Ladd and Butler, 1972; Ross, 1975; Frankenberger and Tabatabai, 1981; Deng and Tabatabai, 1997) suggest that tillage and residue management practices have similar effects on the activity of different enzymes involved in C, P, N, and S cycling in soils. Beare *et al.* (1993) found that microbial growth is favored in the minimum tillage soil environment, resulting in greater enzyme activities.

ii. CARBOHYDRATES. Carbohydrates influence soil quality mainly through their role in the formation and stabilization of aggregates (Cheshire, 1979). Lynch and Bragg (1985) reported that the incorporation of organic residues rather than their removal or burning provides a substrate for soils, which in turn will provide polysaccharide-binding agents. It has also been reported that no-tillage usually increases the carbohydrate content of soils (Arshad et al., 1990; Hu et al., 1995) and these changes can occur within a short period of time. However, ploughing to incorporate organic residues tends to encourage the decomposition of native SOM and thus a reduction in available substrate. The balance between these two opposing occurrences will, to a large extent, determine whether the microbial biomass and aggregation stability would increase or decrease (Haynes and Francis, 1993).

B. RESPONSES OF CROP GROWTH AND YIELD

1. Germination, Seedling Establishment, and Growth

In general, the early removal of straw by burning has been used by farmers to get rid of the straw from the field for better seedbed preparation for the seeding of small seed crops and especially for crops grown for seed production and some cereals (Hamblin *et al.,* 1982; Mason and Fischer, 1986; Chan *et al.,* 1987; Cornish and Lymberg, 1987; Darby and Yeoman, 1994). Since the enforcement or ban on burning in some countries, crop residues are utilized on farm, either incorporated or left on the surface. As direct drilling or minimum tillage leaves the straw from a previous crop on the soil surface without or minimum incorporation, these residues are either mulched or left on the surface as such, depending on the availability of machinery and time with the farmers.

In drier and tropical climates, surface straw has been shown to provide better soil environment by reducing the soil temperature and conserving soil water, resulting in better seedling establishment (Osuji, 1990; Weaich *et al.,* 1996). In cooler and humid climates, however, soil warming and seedling emergence can be delayed in systems that leave high levels of surface residues (Schneider and Gupta, 1985; Hayhoe *et al.,* 1993; Darby and Yeoman, 1994; Burgess *et al.,* 1996; Swanson and Wilhelm, 1996), which potentially affect plant growth and yields. These effects were attributed mainly to a considerable reduction in seed zone soil temperatures (Bidlake *et al.,* 1992; Bussière and Cellier, 1994).

Poor stand establishment in no-tillage surface residues has been related to difficulties in seeding through thick residue mulch (Staniforth, 1982; Felton *et al.,* 1987; dos Santos *et al.,* 1993; Burgess *et al.,* 1996; Swan *et al.,* 1996; Kumar, 1998), and wet soil makes it more difficult for coulters to cut through the residues. Seed was sometimes pushed into the ground, together with the residue, with little or no soil–seed contact (e.g., Weill *et al.,* 1989; Hayhoe *et al.,* 1993). This low soil–seed contact problem becomes more severe under dry soil conditions (Chastain *et al.,* 1995). Shallowly buried straw may impede drilling and causes puffy seed beds with little moisture retention, which can result in poor rooting (Kirkegaard *et al.,* 1994) and patchy establishment. In a high residue situation, trash wheels or trash whippers may be added to the planting unit, taking care to adjust the height so that residue, but not soil, is removed from the seed row [Ontario Ministry of Agriculture and Food (OMAF), 1993]. The removal of residue from a wider strip of row area may benefit plant growth further in some areas by increasing soil temperature in the row (Kaspar *et al.,* 1990; Swan *et al.,* 1996) and minimizes developmental delays (Fortin, 1993; Swan *et al.,* 1996), but the problem remains for cereals that are seeded in narrow rows. Incorporation of crop residues may result in a temporary immobilization of soil mineral N, with a consequent reduction in early seedling growth and development (Rooney *et al.,* 1966; Burgess *et al.,* 1996).

Crop residues retained at the surface or incorporated may produce some phytotoxic allelochemicals, which are known to reduce the germination and seedling growth of crops (White *et al.,* 1989; Chung and Miller, 1995; Weston, 1996). Failures in germination and a reduction in seedling growth have been related to colonization with fungi (Lynch *et al.,* 1981) and other diseases such as take-all (*Gaeumannomyces graminis*), *Rhizoctonia solanii, Phythium spp.* (Cook and Haglund, 1991; Kirkegaard *et al.,* 1994; Smiley *et al.,* 1996), and other unidentified biological effects (Chen *et al.,* 1989). These factors, especially when coupled with other stresses of the environment, including insects and diseases, temperature extremes, nutrient and moisture variables, radiation, and herbicides, often enhance allelochemical production and reduce their degradation, thus increasing the potential for allelopathic interference (Einhellig, 1996).

Reduced root growth in high-strength soils is also responsible for patchy growth and losses in yield under direct drill with surface straw retained in some high-strength soils (Cornish and Lymberg, 1987; Kirkegaard *et al.,* 1994).

A clear understanding of the mechanism causing germination and growth reductions in this system is required to design management strategies to overcome them. Increasing yield under direct drilling may require more disturbance around the seed, such as with sowing points modified with a blade fitted to rip or fracture a slot in the soil 5–10 cm below the seed (Chan and Mead, 1990). These points provide a narrow zone of low soil strength and disturb fungal hyphae without excessive disturbance and have improved growth and yield of direct drilled crops (Chan and Mead, 1990). Management practices should concentrate on reduction in the amount of undecomposed stubble present at sowing and appropriate rotation to control the amount and quality of residues left.

2. Crop Yields

Ideally, crop residue management practices should be selected to enhance crop yields with minimum adverse effect on the environment.

In the last two or three decades, several workers have examined the effect of residue management practices on the harvested yield of the following crop. Results from these experiments are conflicting because of a number of factors involved, including residue quality, management, and edaphic factors and also the health of the previous crop and their complex interactions with various management factors in determining the ultimate crop yield. This indicates that no one residue management system is superior under all conditions. The constraints identified in the crop production system would guide toward the best option. Thus, effort has been made to relate the effects on yield resulting from various crop residue management practices to the constraints in crop production so that the controversy in the literature can be resolved.

Under conditions of optimum fertility, adequate soil water supply, and absence of pests and diseases, grain yields achieved are largely unaffected by the different management practices as shown by several studies (e.g., Biederbeck *et al.,* 1980; Kitur *et al.,* 1984; Undersander and Reiger, 1985; Maurya, 1986; Rasmussen and Rohde, 1988; Wilhelm *et al.,* 1989; Thomas *et al.,* 1990; Njøs and Børrensen, 1991; Prasad and Power, 1991; Rule *et al.,* 1991). Thus, in this situation, pollution and sustainability of the systems are the major considerations.

However, under different environmental and edaphic constraints, different residue management practices have been found to show different grain yield trends. In cropping systems, where the previous crop was infested with pests and disease, treatments with burning of residues generally yielded more grains (e.g., Doran *et al.,* 1984; Jenkyn *et al.,* 1995; Sumner *et al.,* 1995; Prew *et al.,* 1995). Under low soil moisture and high temperatures during the growing season, no-tillage surface mulch provided higher crop yields (e.g., Dao, 1993; Tian *et al.,* 1993b; Lafond *et al.,* 1996) because of more water conservation, reduced evaporation (Hatfield and Prueger, 1996; Prihar *et al.,* 1996), and favorable soil temperatures for favorable root growth (Chaudhary and Prihar, 1974; Maurya and Lal, 1981) with an improvement in biological processes (Hatfield and Prueger, 1996) and enhanced soil N mineralization (Tian *et al.,* 1993b) as compared to residue removal, burning, or incorporation.

Under low winter temperatures, residue burning, removal, or incorporation provides higher crop yields compared with no-till mulch due to surface residues in no-till mulch reducing the seed zone soil temperatures, resulting in poor or delayed germination and reduction in yields (e.g., Schneider and Gupta, 1985; Kaspar *et al.,* 1990; Bidlake *et al.,* 1992; Hayhoe *et al.,* 1993; Bussière and Cellier, 1994; Burgess *et al.,* 1996; Swanson and Wilhelm, 1996).

Where topsoil is susceptible to both water and wind erosion (Geiger *et al.,* 1992; Lafond *et al.,* 1996) and loss of nutrients from the fertile upper soil (Rasmussen and Collins, 1991; Geiger *et al.*, 1992; Larney and Janzen, 1996), no-tillage surface residues and reduced tillage provide better grain yields (Geiger *et al.*, 1992) compared to other methods.

In situations where allelochemicals are likely to be released from residues, residue burning and removal may provide higher grain yields (White *et al.*, 1989; Chung and Miller, 1995; Stirzaker and Bunn, 1996; Weston, 1996).

If high volumes of surface residues are maintained, failure of the seeding machinery results in poor plant populations and a reduction in yields (Burgess *et al.*, 1996; Swan *et al.*, 1996; Kumar, 1998) compared to residue removal, burning, and incorporation. Thus, if proper machinery for incorporating residues is not available, burning or removal of crop residues is the best option (Felton *et al.*, 1987; Burgess *et al.*, 1996). Poor tiller initiation has also been reported to decrease the yields of crops under no-tillage, straw-retained treatments (Rasmussen and Barrow, 1993), whereas the burning of surface residues in no-tillage treatment increased yields.

Poor-quality residues (high C/N ratio, high lignin, and polyphenol content) are known to cause the temporary immobilization of inorganic N (Bhogal *et al.*, 1997; Nicholson *et al.*, 1997), resulting in less N uptake and lower yields (Bahl *et al.*, 1986; Carefoot *et al.*, 1994; Beri *et al.*, 1995). These kinds of residues have a good mulching effect (Tian *et al.*, 1993b) and, in some cases, burning or removal of these residues provides greater yields compared to incorporation. However, burning is only a short-term option and may show adverse effects on crop yields and soil C and N if continued (Rasmussen and Parton, 1994).

Under conditions of high-fertility soils without any limiting constraint or where long-term additions of crop residues have increased the amount of available N, yield increases by the incorporation of crop residues were usually achieved (Wilhelm *et al.*, 1986; Tian *et al.*, 1993a; Dick and Christ, 1995), especially where the crop demand and availability of nutrients from decomposing residues are synchronized (Becker and Ladha, 1997). Otherwise, nutrients may be leached beyond the rooting zone (Francis *et al.*, 1995; Becker and Ladha, 1997; Myers *et al.*, 1997) and cause groundwater pollution.

The effects of various constraints on crop production under variable environments are conceptualized and presented in Fig. 3. Obviously, the choice of the management practices would depend on the identified constraints in the cropping system. Integrated efforts by soil scientists, agronomists, ecologists, plant scientists, environmentalists, and economists are needed to design a system approach for the best choice of crop residue management practices.

3. Biological Nitrogen Fixation

No-tillage and conservation tillage soils have often decreased amounts of soil nitrate compared to cultivated soils (Dowdell and Cannell, 1975; Doran, 1980),

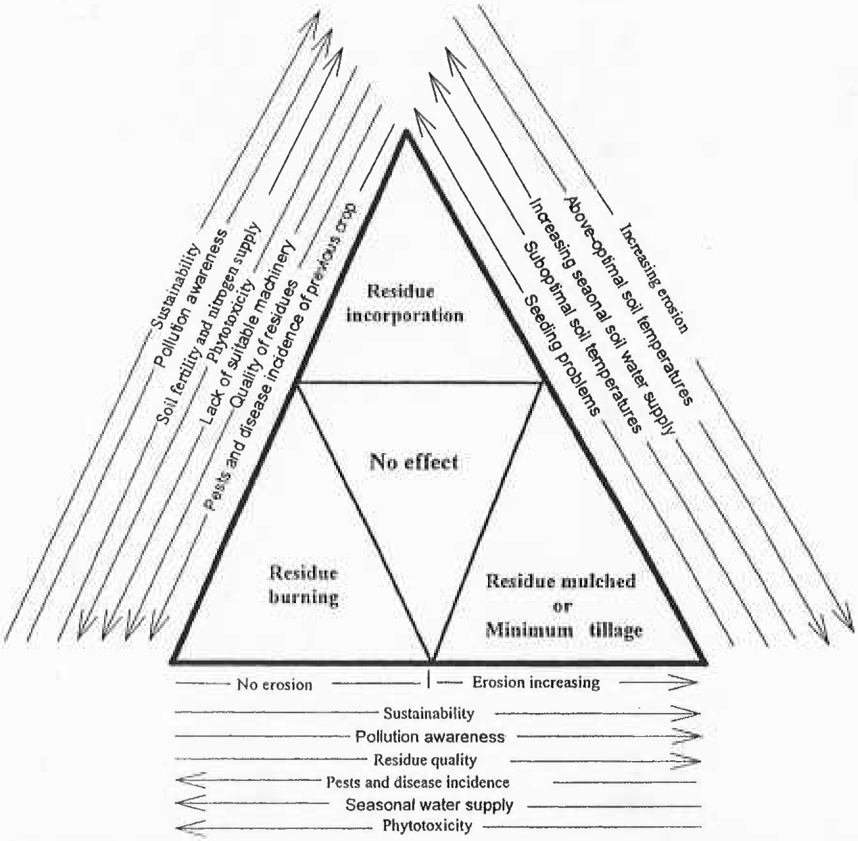


Figure 3 Conceptual diagram for the selection of residue management practice depending on environment and visualized constraints.

which may affect N_2 fixation, as nodulation and N_2 fixation in leguminous crops are increased in soils with a low concentration of nitrates (Bergersen *et al.*, 1989; Cowie *et al.*, 1990). Increased N_2 fixation under direct drilling as compared to cultivated soils has been reported in many leguminous crops (Harper *et al.*, 1989; Hughes and Herridge, 1989; Wheatley *et al.*, 1995). Crop residues with a wide C/N ratio or C-rich sources have also been added with the aim of immobilizing the soil inorganic N and to increase N_2 fixation (Patterson and LaRue, 1983). Wagner and Zapata (1982) found that dry matter yield and N uptake of reference plants were reduced drastically by sucrose addition, but the N in soybean derived from the atmosphere was increased from 30 to 80%. It is, therefore, apparent that this technique can stimulate N_2 fixation by reducing the nitrate concentration in the soil. Similarly, Doughton *et al.* (1993) reported a close inverse relationship between soil nitrate measured at the establishment of chickpea and percentage of N derived from fixation under the wide treatments of prior crop fallowing, N fertilization, or tillage and residue management.

However, in case of field pea, there was no evidence that direct drilling increased N_2 fixation by decreasing the crop uptake of soil N. Because legume species differ in their symbiotic sensitivity to nitrate (Harper and Gibson, 1984), it may not be possible to generalize that direct drilling or incorporation with straw will increase the amount of N_2 fixed by legumes (Evans *et al.,* 1997).

Although the role of crop residues (with wide C/N ratio) in reducing the inorganic N concentration in soil and thus increasing N_2 fixation is well established, in some situations, for example, in acid sandy soils of Niger, west Africa, crop residues addition increased the availability of Mo and P and decreased the concentration of exchangeable Al and Mn, thus enhancing N_2 fixation in *Arachis hypogea* L. (Rebafka *et al.,* 1993).

Another factor to be considered is the release of phytotoxins from decomposing straw inhibiting nodulation and N_2 fixation (Heckman and Kluchinski, 1995) as residues from certain crops and weeds have allelopathic effects and may affect BNF adversely (Guenzi and McCalla, 1966; Freire, 1984). Several earlier studies on allelopathy have demonstrated that the nodulation of legumes may be inhibited by the decomposing residues of other plant species (Rice, 1971; Weston and Putnam, 1985) and this aspect needs to be studied.

4. Phytotoxicity of Crop Residues

The decomposition of some crop residues has been found to have adverse allopathic effects on seed germination, seedling, and crop growth (Elliott *et al.,* 1978; Putnam, 1994) due to substances produced during residue decomposition (Patrick *et al.,* 1963). Microorganisms can produce a large range of substances, potentially toxic to plant roots (Lynch, 1976, as cited by Cannell and Lynch, 1984), but these toxins seldom accumulate in aerobic soils because they are metabolized rapidly by microorganisms.

However, adverse effects of decomposing residues on crops under aerobic conditions have been widely reported. For example, Bhowmik and Doll (1982) reported allelopathic effects from weed residues such as giant foxtail and barnyard grass that affected plant growth and reduced corn and soybean yields. Jessop and Stewart (1983) reported that wheat growth was depressed severely by crop residues (50–70% compared with control), with the ranking of rape residue < pea residue < sorghum < wheat straw. Kimber (1973a,b) also reported a marked depression in wheat germination under wheat straw residue treatments. By eliminating factors such as nutrients, soil structure, and soil pathogens, Stirzaker and Bunn (1996) demonstrated that phytotoxic leachates from the cover crop residues of subterranean clover and ryegrass were responsible for the reduced growth of vegetable seedlings. These workers found that the phytotoxic period of clover ended after 8 weeks in the field and 6 weeks in the laboratory, where conditions favored the decomposition of phytotoxic compounds. The phytotoxic effects of ryegrass lasted longer and were more severe than clover (Stirzaker and Bunn, 1996). Allelopathic effects of rye residue on decomposition on other plant species have also been reported (Rice, 1995; Kessavalou and Walters, 1997).

Vaughan *et al.* (1983) and Nelson (1996) found that phenolic acids, such as ferulic acid, *p*-coumaric acid, and *p*-hydroxybenzaldehyde, released from the living and dead tissue of a variety of plant species caused adverse effects on the growth of crops.

Both mechanical (cultivating the soil to increase aeration) and chemical strategies exist for reducing the phytotoxic effect of crop residues on seedlings. Chemical strategies include the use of amendments such as calcium peroxide, lime formulations (Davies and Davies, 1981; Lynch *et al.,* 1981), and calcium nitrate (Farquharson *et al.,* 1990) to alter the environment near the seeds.

5. Weed Control and Herbicide Efficiency

Crop residues may selectively provide weed suppression through their physical presence on the soil surface as a mulch (Facelli and Pickett, 1991; Teasdale *et al.,* 1991; Moore *et al.,* 1994; Martin, 1996) and by restricting solar radiations reaching below the mulch layer (Facelli and Pickett, 1991), by direct suppression (Dyck and Liebman, 1994), due to allelopathy (White *et al.,* 1989; Weston, 1996), and by controlling N availability (Karssen and Hilhorst, 1992; Seibert and Pearce, 1993).

Residues of rye and other small grains have been shown to inhibit weed emergence and growth (Shilling *et al.,* 1986), probably due to phytotoxic effects. Crutchfield (1985) reported increased weed suppression at increasing mulch rates up to 6.8 tons ha^{-1}. It is apparent that a higher quantity of mulch is required for weed suppression. Even cover crop residues in no-tillage systems have been shown to influence the species composition and population of weeds (Teasdale *et al.,* 1991). Surface crop residues have also been shown to increase the efficiency of some granular herbicides on weed control (Teasdale *et al.,* 1991; Enders and Ahrens, 1995). It is not only the residue placement but also the residue type that influence weed growth in a cropped field (Dastgheib *et al.,* 1999).

In cropping systems where crop yield reductions have been severe due to the competition of weeds for light, water, and nutrients (Kang *et al.,* 1980), the burning of residues is beneficial in removing weeds and weed seeds (Askins, 1991; Rule, 1991; Bowerman, 1995).

In addition to influencing the weed seed reserves in the soil, directly or indirectly controlling weeds, crop residue management and tillage practices also influence the efficiency of soil-applied herbicides (Buhler and Oplinger, 1990). Preemergence herbicides applied must enter the plants from the soil and so their activity might be affected by the presence of straw and ash; consequently, higher rates of preemergence herbicide may be required to control weeds (Beck and Jones, 1996).

In no-till planting systems, the presence of ash from the burning of crop residues reduces herbicide efficiency (Butterworth, 1985; Rule, 1991; Bowerman, 1995), whereas the C in burnt crop residues is highly adsorptive of herbicides and can reduce herbicide performance substantially if such residues are retained close to the soil surface (Bowerman, 1995; Printz *et al.,* 1995). The mulch or stubble from a previous crop present at the time of application of preemergence herbicides not only intercepts herbicides and acts as a physical barrier to prevent the herbicide from reaching the soil (Banks and Robinson, 1982), but also adsorbs the active ingredient of herbicide (Sanford, 1982; Rule, 1991), thus reducing its efficiency. In other occasions, however, the intercepted herbicide can be washed into the soil by rainfall and their efficiency remains high (Johnson *et al.,* 1989).

Although ploughing in of residues has been shown to increase herbicide efficiency (Rule, 1991), the long-term buildup of crop residues can reduce herbicide efficiency (Bowerman, 1995).

However, as discussed earlier, crop residues left on the surface can suppress weed seed germination and or seedling growth and complement the effects of herbicide (Crutchfield *et al.,* 1986). Thus, the overall effect of residues on herbicide performance probably depends on many interacting factors, such as environmental and management factors and amounts of residues remaining in the soil (Mills and Witt, 1989).

6. Pests and Diseases

Wheat yields, like other crops, are generally depressed when planted into a seedbed with residues of previous crop still lying on surface, a problem recognized since the introduction of stubble mulch farming into the North American Great Plains in the 1940s (McCalla and Army, 1961; Smiley *et al.,* 1993). Similar problems have been reported for wheat planted into wheat residues in Australia (de Boer *et al.,* 1993; Burgess *et al.,* 1993), India (Singh *et al.,* 1993), England (Prew *et al.,* 1995), the U.S. Pacific Northwest (Papendick and Miller, 1977), south Africa (Wiltshire and du Preez, 1993), and New Zealand (Cromey, 1996; Fraser and Francis, 1996).

Fungal diseases are the most common and some of these are soilborne or seedborne whereas others are spread by airborne spores and can cause wide spread damage (Gair *et al.,* 1987). The persistence of most of the fungi that affect cereals have a limited host range. Barley, for instance, will not contact mildew or rust from volunteer plants of wheat nor from adjacent wheat crops and vice versa. The unspecified pathogen causing take-all, however, can affect both crops and carryover from one to the other (Gair *et al.,* 1987).

As pests and diseases can affect crops separately or simultaneously, the system becomes very complex with few definitive principles for management. Where a particular disease or pest is a problem, some residue management options may be more suitable than others for further reduction in the incidence of disease or pests and these are reviewed.

The burning of crop residues is thought to be a good option for disease control (Butterworth, 1985; Burgess *et al.,* 1993). Although burning can destroy pests, such as aphids present in residues, the increase in temperature can reduce both harmful and beneficial soil microbes at or near the soil surface (Biederbeck *et al.,* 1980), and even cause shifts in microbial communities (Beare *et al.,* 1996).

The burning of stubble has been shown to reduce the incidence of *F. graminearum* infection of wheat in 2 out of 5 years at one site and in 3 out of 4 years at another site in Australia (Burgess *et al.,* 1993). The failure to control infection in other years was attributed to susceptible weed hosts and poor burning. Effects of burning depend clearly on the burning intensity, degree, and temperatures reached during burning (Singh *et al.,* 1993; Johnston *et al.,* 1996) because even small amounts of inoculum left in the soil may be able to carryover the disease to the next crop (Staniforth, 1982; de Boer *et al.,* 1993). This may be the reason for disease incidence and severity not being reduced in some years by burning (e.g., Rasmussen and Rohde, 1988; Sumner *et al.,* 1995). In some cases, the disease index was not reduced by burning treatments and seedlings grew better compared to residue incorporation and generally yielded more grains (Jenkyn *et al.,* 1995; Sumner *et al.,* 1995).

The effect of incorporating residues on pests and disease incidence are controversial, although Rule *et al.* (1991) found that straw incorporation did not usually exacerbate pests and diseases compared with burning. Take-all lesions in wheat were up to two times higher in conventionally cultivated treatments than in direct drilled treatments (de Boer *et al.,* 1993). Tillage has been shown to have a short-term stimulatory effect, causing increased populations of both applied and indigenous bacteria and indigenous fungi (Donegan *et al.,* 1992). Ploughing was found to cause more severe diseases (eyespot and sharp eyespot) compared to other shallow noninversion cultivation (Jenkyn *et al.,* 1995; Prew *et al.,* 1995). Jenkyn *et al.* (1995) found that the severity of take-all was decreased by ploughing, which was also reported in a long-term study by Prew *et al.* (1995), but only for 2–3 years and after that the severity was increased in the ploughing treatment. A high severity of take-all has also been reported in treatments where wheat straw was incorporated into the soil compared to mulch or burnt (de Boer *et al.,* 1993). Other evidence suggests that the disease was occasionally more severe where residues remained at the soil surface (Gair *et al.,* 1987).

Colbach and Meynard (1995) explained these variations on the basis of crop succession and soil tillage. Where the previous crop was a host crop preceded by a nonhost crop, soil inversion buried host residues, thus decreasing the primary infection risk. Where the previous crop was a nonhost crop preceded by a host crop, soil inversion carried the host residue back to the soil surface, thus increasing the primary infection risk. This phenomenon becomes more obvious in long-term studies (Burgess *et al.*, 1993; de Boer *et al.*, 1993; Prew *et al.*, 1995).

The presence of crop residues resulted in wetter and cooler conditions in the surface soil, thus favoring diseases, and also provides pathogens with an additional source of energy to multiply in number (Cook and Haglund, 1991). There might also be an increased incidence of slugs where residues remain on the soil surface (Butterworth, 1985; Gair *et al.*, 1987).

It has been observed that leaving straw residues on the soil surface increased the number of argentine straw weevil, pysius, springtails, and aphids carried over between crops. The number of actinomycetes, fungi, and algae also might increase in a nonburnt, direct drill situation, which leads to an increase in debris-borne disease such as eyespot, rhynchosporium, brown stem rot, yellow leaf spot, net blotch, and barley scald (Meese *et al.*, 1991; Hermann, 1992). The incidence of yellow cereal fly (*Opomyza florum*) has been reported to be less under mulched treatments compared to burnt and incorporated treatments (Prew *et al.*, 1995).

It is obvious that each crop residue management practice has its drawbacks for different diseases and pests. Thus, decisions on residue management should be made regarding the health of the previous crop and the potential susceptibility of the next crop, cultivar selection, crop rotation, planting date, and plant nutrition (Gair *et al.*, 1987; Smiley *et al.*, 1993).

C. EFFECT OF RESIDUE MANAGEMENT ON THE ENVIRONMENT

Crop residue management and tillage both affect the environment through its influence on losses of soil, plant nutrients, and chemicals to the environment, causing pollution, which is of great public concern. The burning of crop residues produces smoke, which creates hazards on visibility and ash, which pollute the surrounding air if transported by high wind velocities, especially under dry climates and surface waters (Iwamoto *et al.*, 1992; Cihacek *et al.*, 1993; Singer and Warkentin, 1996). For this reason, a burning code was developed in the United Kingdom in 1984 (at that time the burning of residues was allowed) that required the incorporation of ashes within 36 hr of burning (Butterworth, 1985). A burning ban was imposed in Europe (Prew *et al.*, 1995) and the United States (UNEP/FAO, 1977). It is currently being considered in New Zealand (Fraser and Francis, 1996).

In addition, gases released during burning contribute to ozone depletion, acid rain, and the greenhouse effect (Bouwman, 1990). The N-based gases formed during burning include greenhouse gases such as nitrous oxide (N_2O), nitric oxide (NO), and probably also sulfur dioxide (SO_2). Amounts of these gases and their effects have not been determined in many parts of the world, although in the United States it is estimated that agricultural burning accounts for 6% of the total annual emissions of polynuclear aromatic hydrocarbons (Wilbert *et al.*, 1992).

Soluble nutrients present in the ash after burning residues, such as K, NH_4^+, SO_4^{2-}, and, to some extent, Mg, are susceptible to loss from the farming system by leaching below the root zone and can cause groundwater pollution.

Management of crop residues other than by burning may also produce greenhouse gases (Rice *et al.*, 1988; Christensen *et al.*, 1990; Eichner, 1990; Aulakh *et al.*, 1991a,b; Walters *et al.*, 1992; McKenney *et al.*, 1993; Boeckx and Van Cleemput, 1996; Nugroho *et al.*, 1996). The presence of crop residues on soil surface may influence ammonia volatilization by acting as a source of urease (Freney *et al.*, 1992). Ammonia volatilization losses may also occur when crop residues decompose on soil surface (Janzen and McGinn, 1991) and denitrification can be stimulated on their incorporation (Aulakh *et al.*, 1992). If, however, residues are burned on the soil surface, the ash, which is alkaline, can enhance ammonia emissions (Bacon and Freney, 1989).

Crop residues and tillage management also affect the erosion of soil and the associated nutrients and chemicals and also leaching, which may pollute the groundwater or surface waters (Martin *et al.*, 1978; Baker and Laflen, 1982; Addiscot *et al.*, 1991; Ocio *et al.*, 1992; Addiscot and Dexter, 1994; Fermanich *et al.*, 1996). Another important factor to consider is the fuel energy consumed and pollution created by machinery in incorporating the residues. For example, in a desk study in the United Kingdom, Chaman and Cope (1994) found that mould board ploughing to a depth of 20 cm rather than tine and disc cultivation to a depth of 10 cm increased diesel consumption by 15 Mliters per year in England and Wales. This represented 734×10^{12} J per year of fossil fuel energy consumption and emissions of 807×10^3 kg NO_2-N, 139×10^3 kg particulates, and 40×10^6 kg CO_2 per year.

V. SOIL NITROGEN DYNAMICS AND CROP NITROGEN RECOVERY

A. Nitrogen Mineralization/Immobilization Turnover

Decomposition and mineralization are the means by which nutrients either held in the SOM or added through organic materials (manures, crop residues) are released into the soil as inorganic forms. The inorganic N released becomes available either for subsequent recycling and utilization by plant or microorganisms or is lost from the system. Soil N mineralization is the transformation process where ammoniacal or ammonium N (NH_4^+) or ammonia (NH_3) is released by soil microorganisms as they utilize organic N compounds as an energy source (Jansson and Persson, 1982; Jensen, 1997). The process is complex and depends on the activity of nonspecific heterotrophic soil microorganisms under both aerobic and anaerobic conditions (Jarvis *et al.,* 1996). Both the magnitude and the rate of mineralization are different for newly added residue and existing, already degraded organic materials of varying ages and degrees of recalcitrance.

Soil N mineralization is always accompanied by N immobilization (Fig. 4); the processes are intimately connected and dependent. Much of the NH_4^+ or NO_3^- or simple organic N compounds that are released are assimilated by the soil microbial population and transformed into the organic N constituents of their cells during the oxidation of suitable C substrates through the process of immobilization. However, immobilized N is likely to be available subsequently for mineralization as the microbial population turnover. The degradation of microbial tissue 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 parts of mobile N. Concurrent with the release of N from the soil microbial biomass there will also be direct release from fresh residues and "native" soil organic materials of various ages. Nitrogen continually cycles between inorganic and organic phases via the mineralization/immobilization activities of soil microbes,

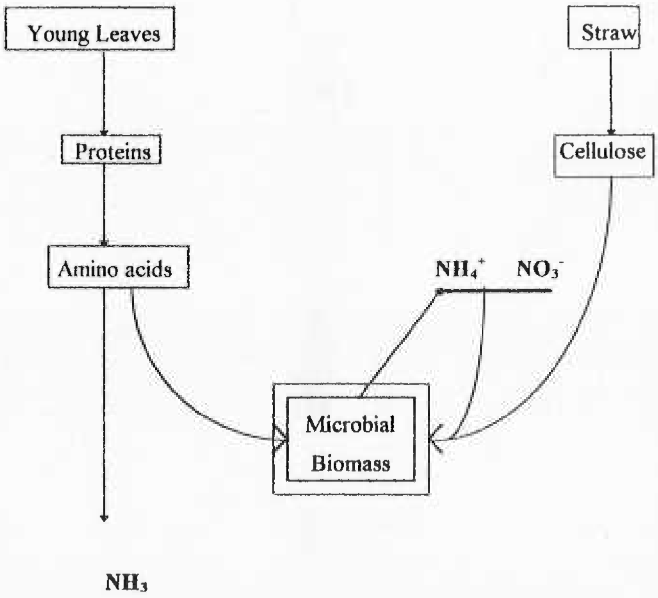


Figure 4 Mineralization of organic matter incorporated into soils. Nitrogen-rich matter (e.g., young leaves) produces inorganic nitrogen, whereas matter poor in nitrogen (e.g., straw) consumes nitrogen (immobilization). Both processes feed the microbial pool. From Mengel (1996).

hence termed the biological turnover or mineralization–immobilization turnover (MIT) (Jansson, 1971). Because these processes are concurrent but in opposite directions, net immobilization or net mineralization is often used to indicate the dominant process. In general, net immobilization occurs when the soil microbial population expands due to a substrate addition and N demand exceeds the supply from the soil or fertilizer. Alternatively, net mineralization results when the residues of inorganic N exceeds the demands of soil microorganisms as during the decline of a microbial population.

A basic assumption of MIT is that all immobilization occurs from the inorganic pool. Ammoniacal N has been shown to be preferred (Jansson, 1958; Recous *et al.*, 1988). However, where NH_4^+ is not available, NO_3^- is assimilated by the soil microbial biomass 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). 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 "direct hypothesis." Although it has been demonstrated that the soil microbial biomass 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). The way in which the mineralization/immobilization process operates is important in the turnover, recycling, and fate of released N from the added fertilizer, crop residues, and native SOM.

1. Involving Fertilizer Nitrogen

Field experiments that include plots with and without fertilizer N enable the "apparent" recovery of the fertilizer N to be calculated from the amounts of N in harvested herbage. The retention of N in stubble and roots is generally ignored in the calculation of recovery, with some justifications, as the effect of fertilizer N on the amounts of N in roots and stubble is often relatively small. The "apparent" recovery is defined as difference between fertilized and unfertilized plots in the amounts of N harvested in aboveground parts, usually at the harvest of arable crops, expressed as a percentage of the amount of N applied in the fertilizer. On this basis, the "apparent" recovery of fertilizer N by arable crops is generally in the range of 40–60%. For example, Grove (1979) showed that the recovery of fertilizer N by maize was similar in temperate and tropical regions. At N application rates between 35 and 120 kg N ha^{-1}, a fairly constant 55% was recovered in harvested plant tops. Nitrogen recovery decreased quickly when application rates exceeded the assimilative capacity of the crop so that less than 40% was recovered at the 200-kg N ha^{-1} rate (Grove, 1979).

The advent of ^{15}N tracer methodology made it possible to account for all added forms of N to soil and to distinguish between soil and fertilizer sources. Some of the first findings with ^{15}N revealed higher unlabeled soil N recovery by fertilized plants than unfertilized plants (Broadbent, 1965; Westerman and Kurtz, 1973; Barraclough *et al.,* 1985; Porter *et al.,* 1996). For example, Powlson *et al.* (1992) showed in their experiments on winter wheat at different sites in eastern England on three soils that the application of labeled fertilizer N tended to increase the uptake of unlabeled soil N by 10 to 20 kg ha^{-1} compared to control receiving no fertilizer. This was probably due to pool substitution (i.e., labeled inorganic N standing proxy for unlabeled inorganic N that would otherwise have been immobilized or denitrified). These data implied that N fertilization stimulated the mineralization of native soil organic N (SON). This phenomenon, which has been referred to in the literature as the "priming effect" or "added N interaction" (ANI) (Hauck and Bremner, 1976; Azam *et al.,* 1985; Jenkinson *et al.,* 1985), had been discussed earlier, when research with green manures showed enhanced mineralization of soil N (Lohnis, 1926; Bingeman *et al.,* 1953). However, confirming data were not available until ^{15}N was used. As a result, the basic assumption of the difference method that N immobilization and mineralization processes were similar between fertilized and unfertilized soils (Pomares-Garcia and Pratt, 1978) appeared to be invalid.

Other mechanisms have also been proposed to account for ANIs such as osmotic and salt effects (Broadbent and Nakashima, 1971; Westerman and Tucker, 1974), stimulation of rhizosphere microorganisms (Legg and Allison, 1967), stimulation of microorganisms (Westerman and Kurtz, 1973), increased metabolism, and greater soil exploration by expanding root systems accounted for the higher levels of soil N in fertilized plants (Sapozhnikov *et al.,* 1968) and protonation of organic nitrogenous bases (Laura, 1975).

Another finding of ^{15}N fertilizer efficiency experiments was that plant recoveries of applied N were generally lower than those obtained by the difference method (Allison, 1955; Westerman and Kurtz, 1973). Vlek and Fillery (1984) found this disparity especially true in high N paddy soils, although a greater agreement between methods was noted in low N soils. In a study at seven sites in the United Kingdom using wheat (applied with 32.4- to 233.9-kg labeled N ha^{-1}), Powlson *et al.* (1992) reported that crop recoveries of fertilizer N by ^{15}N methods ranged from 46 to 87% (mean 68%), whereas recoveries ranged from 30 to 96% (mean 74.3%) by the difference method. Thus, these workers concluded that from -5 to 63% (mean 16.5% of N uptake by control crop) more unlabeled soil N was taken up by the crop when ^{15}N-labeled fertilizer was added. While several researchers consider these differences due to priming effects, another explanation postulates that they are caused by isotope exchange, resulting from interactions with soil microorganisms.

An alternative explanation for the disparity between the difference method and the ^{15}N method, as well as the apparent mineralization of soil organic N, was initially proposed by Jansson in 1958. He postulated that the biological interchange between added N and soil microorganisms was a significant factor controlling ^{15}N recovery. Further research is necessary to quantify the extent of microbial N processes under the various management conditions that are imposed in agricultural systems, which ultimately control the recoveries of applied fertilizer N by crops.

2. Involving Crop and Organic Residues

The application of crop and organic residues to soil involves a substantial input of carbonaceous material and thus may result in the immobilization, at least temporarily, of some inorganic N already present in the soil. However, the balance between immobilization and mineralization changes with time and, in the long run, increasingly favors mineralization. Thus, the C/N ratio has long been known to be of great relevance to the rate with which N is released from crop residues (Jensen, 1929; Ford *et al.,* 1989; Quemada and Cabrera, 1995; Janssen, 1996; Whitmore and Handayanto, 1997; Fig. 5). As the influence of the crop residue type and the related quality (C, N, or lignin content, C/N and lignin/N ratios) on residue decomposition have already been discussed, their influence on mineralization of N is reviewed briefly here.

The addition of plant residues most often results in a net N immobilization phase followed by a net remineralization phase (as evaluated by the difference between

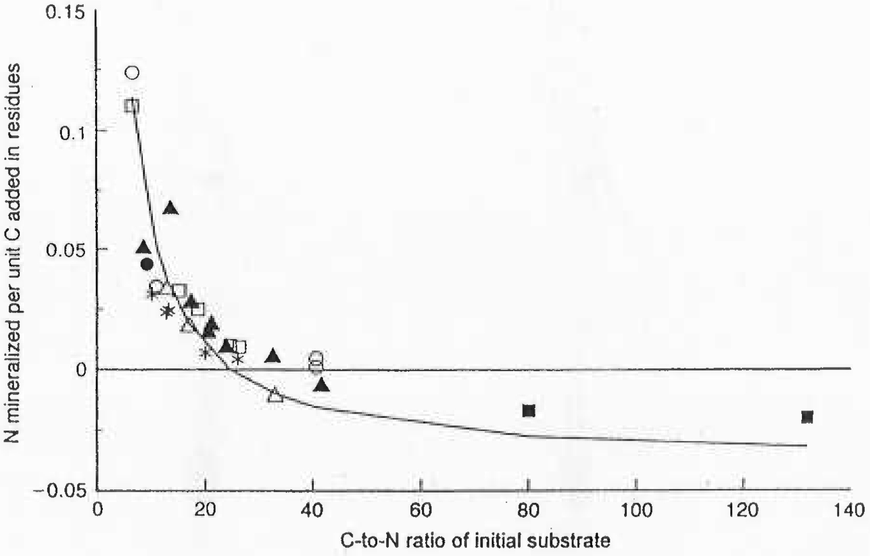


Figure 5 Relationship between nitrogen mineralized (or immobilized) and the carbon/nitrogen ratio of added organic matter. *, Jensen (1929); ▲, Chae and Tabatabai (1985); ■, Nieder and Richter (1989); □, Franzluebbers *et al.* (1994); △, Zagal and Persson (1994); ●, Thorup-Kristensen (1994); ○, A. P. Whitmore, unpublished. The solid line represents theoretical relationship [Eq. (6)]. From Whitmore and Handayanto (1997).

an amended soil and a control soil). The dynamics as well as the net amounts of N immobilized varied greatly in these experiments according to the nature of plant residues.

Harmsen and van Schrevan (1955) reported that crop residues with a C/N ratio below 30 are expected to result in net mineralization, whereas a C/N ratio wider than 30 favors immobilization. Das *et al.* (1993) reported that sorghum straw with a C/N ratio of 72 resulted in the immobilization of N up to 90 days. According to Stevenson (1986), net immobilization lasts until the C/N ratio of the decomposed material has been lowered to about 20. However, as in earlier studies (Azam *et al.*, 1993), net immobilization of N has been reported to take place during early decomposition with a residue C/N ratio as low as 15 (Jensen, 1994a, 1997), probably due to C/N ratios of readily decomposable material being different from the overall C/N ratio of the material. In the study of Quemada and Cabrera (1995), the C/N ratio of the organic materials remaining on the soil surface at the end of experiment for three cereal stems was >28, which indicates that the N mineralization can commence before the C/N ratio of the residue is lowered to 20.

Results of Aulakh *et al.* (1991a) supported the observation of Smith and Sharpley (1990) of less drastic effects on soil N immobilization when high C/N ratio crop residues were left on the soil surface than when they were incorporated. The reason was that more N was immobilized by incorporated residues. Greater amounts of fertilizer ^{15}N were found by Cogle *et al.* (1987) in incorporated straw than in surface straw in field studies, further implicating incorporated straw as immobilizing more of applied or soil N.

Tracing soil mineral N and/or residue N pools with ^{15}N has been used to measure variations in N and ^{15}N mineral pools or in N and ^{15}N organic pools (soil biomass) by using calculations based on the isotopic dilution technique (Barraclough, 1991). This has been used to describe the dynamics of gross N mineralization and immobilization after residue incorporation (Sorensen, 1981; Ocio *et al.*, 1991; Jensen, 1994a; Recous *et al.*, 1995; Watkins and Barraclough, 1996). A similar approach has been adopted using combined treatments identical in the total amount of added N and C but differing only in the N pool being labeled (^{15}N residues + ^{14}N mineral and ^{14}N residues + ^{15}N mineral) (Mary and Recous, 1994).

Once the residue N is mineralized, it is taken up by plants (Jordan *et al.*, 1993), recycled in the microbial biomass for their growth (Mary *et al.*, 1996; Jensen, 1997), stabilized in complex soil organic matter (Jansson and Persson, 1988), or lost from the soil plant system (Harper *et al.*, 1987; Aulakh *et al.*, 1991a,b; McKenney *et al.*, 1993; Haynes, 1997). Finally, this mineralized N would benefit the subsequent crops.

B. CROP NITROGEN RECOVERY

The recovery of mineralized N by a subsequent crop from either plant residues or fertilizer is the product of net mineralization and the efficiency with which inorganic N is assimilated by a subsequent crop. The efficiency of uptake is similar for high N leguminous residues and that of fertilizers, but for high C/N ratio crop residues it is slightly lower (Janzen, 1990; Bremner and van Kessel, 1992). This efficiency depends largely on the temporal patterns of net mineralization, plant N uptake, and N losses.

1. Fertilizer Nitrogen by Crops

Recoveries as high as 73–80% of applied fertilizer N by sorghum and wheat (tops + roots) and 61–66% (tops only), have been reported (Haynes, 1994; Jordan *et al.,* 1996). The range of reported values of N recovered by subsequent crops from other studies is presented in Table V. In general, between 20 and 87% of applied N is recovered by the first crop and between 10 and 35% is retained in the soil and 1–35% is unaccounted for (Table V). The N that is retained in soil is probably immobilized and enters the soil organic N pool. The reported variation in crop recoveries may be due to different fertilizer sources used in different studies, different rates of fertilizers used, climate, management practices, and the test crop grown. The fixation by OM and clay lattices can effectively lower the availability of N to plants (Nommik, 1965). Another possibility for the poor plant recovery of fertilizer N is the competition between plants and soil microorganisms. The incorporation of N into a microbial biomass through the immobilization process essentially removes N from the plant available pool. Several studies have identified biological immobilization as having a significant role in controlling plant N availability (Ladd and Amato, 1986; Recous *et al.,* 1988; Haynes, 1997).

One might view net immobilization as a way of storing N for future crops. However, recoveries of residual fertilizer N in subsequent crops have been quite limited (Legg and Alison, 1967; Thomsen and Jensen, 1994). For example, Jannson (1963) found that only 1% of immobilized N would be remineralized per year. In other studies, between 1 and 10% recovery in second-year crops has been reported (Table V). Hart *et al.* (1993) found that only 16% of the labeled ^{15}N remaining in the soil (0–70 cm) and stubble in the year of application was taken up by subsequent crops during 4 residual years, 29% was lost from the soil/crop system, and 55% remained in the soil. Thus, added N seems to undergo stabilization and modification to less active forms. In studying SOM dynamics and transformation, it is necessary to understand the short- and long-term effects on plant recovery of applied N.

Several pools and pathways have been delineated to describe the fate of unrecovered N, which is often considered as lost from the system through various edaphic loss processes such as leaching, denitrification, and ammonia volatilization (Ladd and Amato, 1986; Aulakh *et al.,* 1992; Porter *et al.,* 1996; Haynes, 1997). Another source of loss is from the plant itself by ammonia volatilization from leaves and root exudation (Harper *et al.,* 1987; Papakosta and Gagianas, 1991).

Table V

Recovery, Retention, and Estimated Losses of Fertilizer N Added to Different Crops Reported by Different Workers

N source/ fertilizer	N rate applied (kg ha^{-1})	Crop	N recovered in crop (%)		N retained in soil (%)	Estimated N loss from system[a] (%)	Reference
			Year 1	Year 2	Year 1[b]	Year 1[b]	
Urea	52	Wheat	35.1	—	—	—	Palta and Fillery (1993)
$(NH_4)_2SO_4$	225	Winter Wheat	55	—	27	24	Powlson (1993)
	240	Oilseed rape	48	—	26	23	Powlson (1993)
	220	Potatoes	53	—	22	25	Powlson (1993)
$(NH_4)_2SO_4$	30	Spring Wheat	23–34	—	—	—	Bremner and van Kessel (1992)
NH_4NO_3	100	Barley	34.5	2.2	—	—	Thomsen and Jensen (1994)
	100	Barley	40.1	—	—	—	Thomsen and Jensen (1994)
	100	Barley	29.3	—	—	—	Thomsen and Jensen (1994)
$(NH_4)_2SO_4$	56	Corn	45	—	—	—	Hesterman *et al.* (1987)
	168	Corn	49	—	—	—	Hesterman *et al.* (1987)
$(NH_4)_2SO_4$	100	Sugarbeet	43–46	1	26–29	25–31	Zapata and van Cleemput (1986)
	50	Winter rye	20–27	2	—	—	Zapata and van Cleemput (1986)
NH_4NO_3/KNO_3	150–225	Winter wheat	46–87	—	7–14	1–35	Powlson *et al.* (1992)
KNO_3	50	Rye monoculture	39	8–10	36	25	Ranells and Wagger (1997)
	50	Rye-clover	19	—	40	41	Ranells and Wagger (1997)
	50	Clover	4	—	32	64	Ranells and Wagger (1997)

[a] Unaccounted for ^{15}N was taken to be as N loss from the system.

[b] No corresponding data were available for year 2.

2. Legume and Nonlegume Nitrogen by Succeeding Cereal Crops

Most estimates of the N benefit of legumes are based on the use of added ^{15}N-labeled legume residues to the soil (Ladd and Amato, 1986) and not on *in situ* residues left in the field as this has not been conducted extensively. Experiments in Australia (Ladd *et al.,* 1981,1983; Ladd and Amato, 1986; Muller and Sundman, 1988; Armstrong *et al.,* 1997), Canada (Janzen *et al.,* 1990; Bremner and van Kessel, 1992), and the United States (Harris and Hesterman, 1990; Harris *et al.,* 1994) have shown that cereals recovered between 10 and 34% of the ^{15}N applied in legume residues (Table VI). In some cases, 50–70% of the N applied as soybean and lucerne residues has been recovered by corn (Hesterman *et al.,* 1987).

In the Philippines, Morris *et al.* (1986a,b) showed that in short-term green manures, 33–49% of the N applied in residues of mung bean and cowpeas was taken up by a rice crop. Only a few studies have been conducted on the recovery of pasture N by cereals. In one study, wheat was found to recover approximately 10% of N from a ryegrass/white clover (70:30, w/w) pasture (Haynes, 1997). Legume roots were found to contribute significantly toward the nutrition of subsequent cereals (Sawastsky and Soper, 1991; Thomsen *et al.,* 1996).

Nitrogen recoveries from nonleguminous residues (Table VII) are even lower than that from leguminous residues (3–18% vs 10–34%; Table VI), except in one case, where the N recovery by a rice crop was reported to be 37% from wheat straw. The reason is because of the low C/N ratio (34) of the wheat straw and the submerged conditions (Norman *et al.,* 1990).

Although the limited plant availability of legume N is primarily due to the stabilization of N in soil organic forms, losses of legume ^{15}N through ammonia volatilization, denitrification, or leaching may also be considerable. For example, Ladd and Amato (1986), Janzen *et al.* (1990), and Haynes (1997) reported that losses of legume ^{15}N were equal or greater than N uptake by a subsequent crop. In general, 39–70% of leguminous residue N (Table VI) and 54–81% of nonleguminous residue N (Table VII) were retained in soil. The unaccounted N varied from 20 to 50% of leguminous residue N (Table VII) and only 9–16% of nonleguminous residue N (Table VII). In the case of nonleguminous residues, limited or slow decomposition and stabilization of N in organic forms may be the reason for low N availability to subsequent crops (Bremner and van Kessel, 1992). In the second and third years, only 1–5% of the leguminous or nonleguminous residue N was recovered in crops (Ladd and Amato, 1986; Ta and Faris, 1990; Thomsen and Jensen, 1994; Haynes, 1997).

3. Comparison of Nitrogen Recovery from Applied Fertilizer and Legume Nitrogen

In general, several studies have shown that the recovery of ^{15}N from labeled leguminous and nonleguminous crop residues by subsequent cereal crops was

Table VI

Recovery, Retention, and Estimated Losses of Leguminous Residue Nitrogen Added to Different Crops Reported by Different Workers

Residues added	Crop	N recovered in crop (%)		N retained in soil (%)		Estimated loss from system[a] (%)		Reference
		Year 1	Year 2	Year 1		Year 1		
Pisum sativum (C/N = 14–16)	Barley	11	4.3	—	—	—	—	Jensen (1996)
T. repens; V. faba; T. Subterraneum (% N = 1.6–3.0)	Barley	6–25	—	—	—	—	—	Muller and Sundman (1988)
Glycine max (C/N = 15)	Rice	11	—	39	—	50	—	Norman *et al.* (1990)
L. culinaris green manure (% N = 4.05)	Wheat	19	—	—	—	—	—	Bremner and van Kessel (1992)
L. culinaris straw	Wheat	5.5	—	—	—	—	—	Bremner and van Kessel (1992)
M. littoralis (% N = 3.2)	Wheat	22–28	3–4	56–70	—	3–20	—	Ladd *et al.* (1983)
T. pratense L.	Corn	15	—	57	—	27	—	Harris *et al.* (1994)
M. littoralis L. (% N = 3.2)	Wheat	16–19	4–5	—	—	—	—	Ladd and Amato (1986)
G. max								
Roots	Oats	0.53	—	—	—	—	—	Bergersen *et al.* (1992)
Trash	Oats	9.85						Bergersen *et al.* (1992)
Leaves	Oats	18.17						Bergersen *et al.* (1992)
M. sativa (% N = 2.6)	Barley	11	4	—	—	—	—	Ta and Faris (1990)
L. perenne/T. repens (70:30; C/N = 18)	Winter wheat	9.9	1.5	60(5.2)[b]	55	25	10	Haynes (1997)
	Spring wheat	9.0	1.5	60(5.8)	55	24	14	Haynes (1997)

[a] Unaccounted for ^{15}N was taken to be as N loss from the system.

[b] Recovered in undecomposed residues.

Table VII

Recovery, Retention, and Estimated Losses of Nonleguminous N Added to Different Crops Reported by Different Workers

Residues added	Crop	N recovered in crop (%)		N retained in soil (%)	Estimated loss from system[b] (%)	Reference
		Year 1	Year 2	Year 1[a]	Year 1[a]	
Wheat straw (C/N = 34)	Rice	37	—	54	9	Norman *et al.* (1990)
Rice straw (C/N = 59)		3	—	81	16	Norman *et al.* (1990)
Wheat straw (C/N = 43)	Wheat	5.5	—	—	—	Bremner and van Kessel (1992)
Barley straw (N% = 0.5)	Barley	4.5	2.6	—	—	Thomsen and Jensen (1994)
Barley straw + ryegrass cover crop	Barley	4.4	2.7			Thomsen and Jensen (1994)
Ryegrass	Barley	10.2	2.4			Thomsen and Jensen (1994)
Ryegrass + barley straw	Barley	7.8	2.1			Thomsen and Jensen (1994)
Sorghum (C/N = 20–44)	Sorghum/barley	4.5–25	<5	—	—	Vigil *et al.* (1991)
Barley straw	Barley/mustard	8	3.3	—	—	Jensen (1996)
Wheat straw	Sorghum	18–20.6	—	—	—	Jordan *et al.* (1996)

[a] No corresponding data were available for year 2.

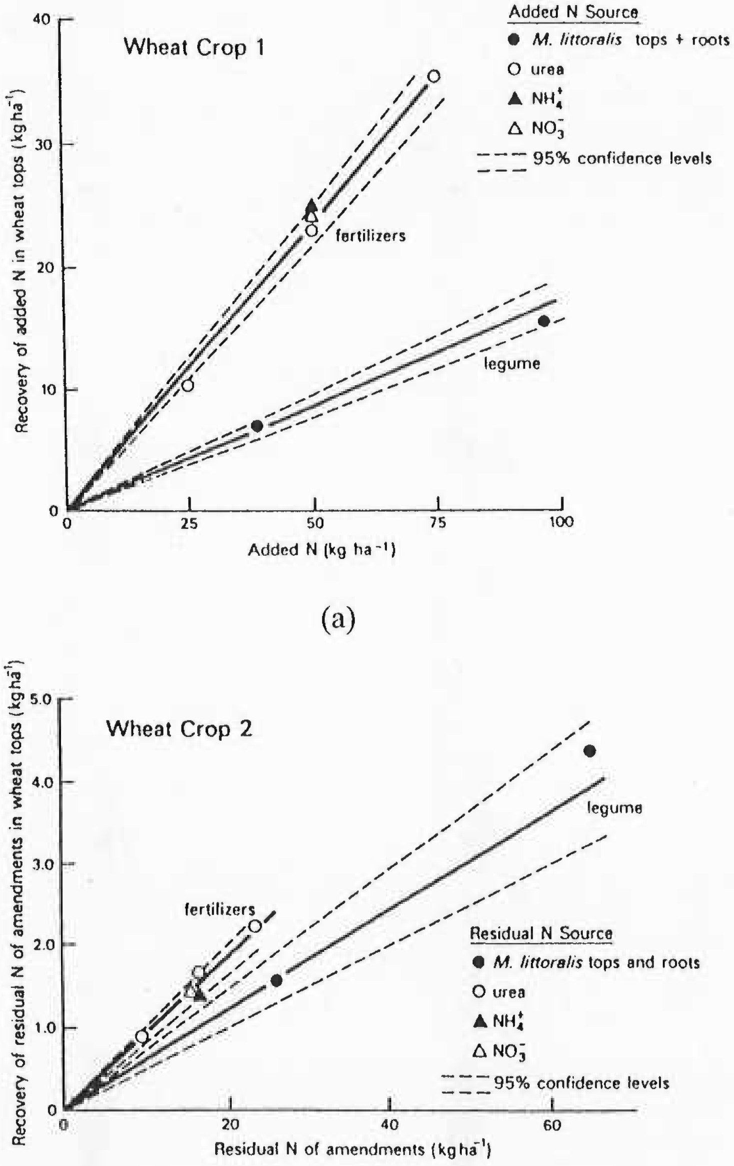
[b] Unaccounted for ^{15}N was taken to be as N loss from the system.

about one-half and one-eighth, respectively, of that from various forms of labeled fertilizer N (Tables V–VII). For example, in Australian dryland, Ladd and Amato (1986) found that the recovery of ^{15}N by the first wheat crop averaged 17 and 46% from legume and fertilizer sources, respectively, after 1 year (Fig. 6a). Subsequent recoveries of both fertilizer and legume N in second year were lower (<5%, Fig. 6b). Total ^{15}N recoveries in plant and soil from fertilizer and legume were 84 and 80%, respectively, indicating similar losses of N from both sources.

Associated measurements revealed that more ^{15}N was immobilized into soil organic N from legume ^{15}N than from fertilizer N. In a greenhouse study, Jordan *et al.* (1996) found that the sorghum crop recovered within 8 weeks 61, 22, and 18% of applied N from fertilizer, clover, and wheat residues, respectively. Likewise, Harris *et al.* (1994) found that more fertilizer than legume N was recovered by crops (40% vs 17% of input), more legume than fertilizer N was retained in soil (47% vs 17% of input), and similar amounts of N from both sources were lost from the cropping system (39% of input) over a 2-year period. More fertilizer than legume N was lost during the first year of application (38% vs 18% of input), but in the second year, more legume N was lost compared to fertilizer N. In contrast to dryland experiments, legume N was found to be as good as fertilizer N based on recoveries by rice crop (Sisworo *et al.,* 1990). It is clear that the synchronization between N mineralization and N uptake is an important factor controlling the recovery of applied N and extent to which mineralized N is lost from the system (McGill and Myers, 1987; Myers *et al.,* 1997; Becker and Ladha, 1997; Haynes, 1997).

4. A Conceptual Approach to Nitrogen Mineralization from Crop Residues

Key processes involved in the soil–plant N cycle are the decomposition of native SOM and plant residues and litter and the accompanying mineralization and immobilization of inorganic N. Even though these processes are complex, considerable advances have been made. It has been shown that crop residues decompose in two distinct phases: an initial more rapid phase, in which about 70% of the C initially present in the residues is lost as CO_2, followed by a slower phase (Jenkinson, 1977; Parton *et al.,* 1987; Xu and Juma, 1995). These two phases represent the labile C (or decomposable) and recalcitrant C (or resistant) fractions of the crop residues (Jenkinson and Rayner, 1977; Van Veen and Paul, 1981; Hansen *et al.,* 1991). However, with regard to N mineralization, the results are often contradictory. A clear two-phase mineralization suggesting a labile and resistant fraction similar to that observed for C has been reported in some studies (Broadbent and Nakashima, 1971); in other studies, this two-phase pattern is not observed (Amato *et al.,* 1984).



(b)

Figure 6 Relationship between the amount of nitrogen added to soil as legume and fertlizer sources and that (a) taken up by a first wheat crop and (b) recovered in cropped soil and that taken up by a second wheat crop. From Ladd and Amato (1986).

In many studies, the C/N ratio, the N concentration of residues, or both have been used as predictors of net N mineralization (Vigil et al., 1991; Quemada *et al.*, 1997). Whitmore and Handayanto (1997) reported that N mineralized can be predicted for a uniformly decomposable residue using Eq. (6) (Fig. 5) as

$$N = C_o\{1/Z - E/Y\}, \tag{6}$$

where N is the amount of N mineralized once decomposition is almost complete, Z is the C/N ratio of the substrate (assumed constant), E is a microbiological efficiency factor that can vary for several reasons, but can be taken as about 0.4, and Y is the C/N ratio of the end product of the decomposition process; humus with a C/N of 10 and C_o is the release of N at the start of mineralization (i.e, at zero time, t_o). Replacing C_o with $(C_o - C_t)$ gives the release of N after time "t".

Microorganisms require N for their growth in order to decompose crop residue, and they and their products must maintain a certain C/N ratio. This value is mostly narrower than the residue themselves, but because most microbes are not 100% efficient ($0 < E < 1$), the critical C/N ratio of 20, where mineralization switches to immobilization, is often wider than the residues and, with the values assumed for E and Y earlier, is about 25, so it is more of a process controlled by supply and demand than decomposition (Whitmore and Handayanto, 1997). This equation may not hold good for residues that are heterogeneous in terms of quality, and results may deviate from what is predicted by Eq. (6) (Fog, 1988). Nevertheless, Whitmore and Handayanto (1997), using data taken from a number of field and laboratory experiments (Jensen, 1929; Chae and Tabatabai, 1985; Nieder and Richter, 1989; Franzluebbers *et al.,* 1994; Thorup-Kristensen, 1994; Zagal and Persson, 1994; Whitmore and Handayanto, 1997), have shown that this Eq. (6) is quite general and holds well. This also forms the foundation of the mineralization sections of many other models (e.g., CENTURY, Paustian *et al.,* 1997; DAISY, Magid *et al.,* 1997b); as it expresses the release of N after decomposition is complete.

Estimates of N mineralization can be made experimentally with a large number of diverse incubation and chemical extractants (Goh and Haynes, 1986). A few of the common predictors are listed in Table VIII. In many other studies, predictors such as the lignin-to-N ratio and N-to-polyphenol ratios appeared more reliable indicators of N mineralization (Table VIII). Berg and Staff (1981) reviewed a number of results and concluded that net N mineralization started at a wide range of C/N ratios. In actual field conditions, this confusion arises because only net N mineralization was measured. Thus the relationships being studied are only between C/N ratios of the organic substrate and a group of soil processes, not the N mineralization alone. Changes in processes such as N immobilization and N losses (e.g., due to denitrification) may blur or completely obscure any postulated relationships between N mineralization and a factor such as the C/N ratio of the residues.

Table VIII

Chemical Predictors Used by Different Workers for Estimating N Mineralization from Crop Residues

Predictor	Residue type	Reference
Hot water extractable N (60 °C, 30 min)	Medic	Amato *et al.* (1984)
6 *N* HCl hydrolysable N	Corn and wheat straw	Martin *et al.*. (1980)
6 *N* H_2SO_4 hydrolysable N	Wheat straw and roots	Jawson and Elliott (1986)
N and C/N ratio	Agroforestry species	Vanlauwe *et al.* (1996)
Polyphenols (Folin–Denin method)	Legumes	Anderson and Ingram (1989)
Lignin concentration	Field crops	Muller *et al.* (1988)
Polyphenol:N ratio	Tropical legumes	Palm and Sanchez (1991)
(Lignin+polyphenol): N ratio	Tropical legumes	Fox *et al.* (1990)
N and lignin: N ratio	Sorghum	Vigil *et al.* (1991)

VI. NITROGEN BENEFITS TO SUBSEQUENT CROPS

Beneficial effects of legumes on the yield of subsequent crops have been demonstrated in many studies (Senaratne and Hardarson, 1988; Jensen, 1994a; Hossain *et al.*, 1996a,b; Armstrong *et al.*, 1997; Holford and Crocker, 1997). This residual effect was noted when the legumes were incorporated as crop residues (Ladd and Amato, 1986; Senaratne and Hardarson, 1988; Haynes, 1997), as green manures (Yadwinder-Singh *et al.*, 1991) grazed by animals (He *et al.*, 1994; Whitehead, 1995), and harvested for hay (Papastylianou, 1987) or for grains (Blumenthal *et al.*, 1988; Hossain *et al.*, 1996a; Armstrong *et al.*, 1997; Holford and Crocker, 1997).

The N benefits in legume-cereal rotations have been attributed entirely to the transfer of BNF (Munyinda *et al.*, 1988). These benefits have also been explained by a greater immobilization of nitrate during the decomposition of cereal compared with leguminous residues (Green and Blackmer, 1995). Others have expressed the view that N benefits may be due to a combination of legume N sparing and the transfer of fixed N_2 (Keatinge *et al.*, 1988; Herridge *et al.*, 1995).

Thus, the beneficial residual effect is not necessarily due to the direct contribution of nutrients from aboveground materials being returned to soil, although the magnitude of the yield increase of the subsequent crop is related to the amount of material returned to the soil. For these residual effects to occur after legumes compared to nonlegumes, it is expected that the amount of fixed N returned by legumes to the soil should be greater than the amount of soil N removed in the harvested grain (Eaglesham *et al.*, 1982; Haynes *et al.*, 1993).

There is no agreement as to whether the beneficial effect of legume to subsequent crops is due to a direct N contribution from N_2 fixation or is a net "rotation contribution," which is a complex effect combining N availability, disease control, and soil structure improvement (Papastylianou and Puckridge, 1983; Dalal *et al.*, 1991; Janzen and Shaalje, 1992; Holford and Crocker, 1997). These effects may include reduced diseases and insects problems, residual treatment effects, and unexplained effects. The use of ^{15}N labeling in recent experiments has permitted some degree of separation of residual and legume N effects (Prasad and Power, 1991; Chalk, 1998).

A. GRAIN YIELD AND NITROGEN RESPONSES

Many cropping experiments have shown increasing yields and N uptake by cereal grown after legumes than when grown after cereals (Strong *et al.*, 1986b). For example, a summary of the effects of the inclusion of lupin, field pea, faba bean, and chickpea in cereal cropping systems in over 30 Australian experiments showed responses from subsequent crops of between 12 and 164% N or 0.18 and 2.28 tons ha^{-1} increase in yield responses (Evans and Herridge, 1987). Likewise, benefits of legumes in the range of 0.2–3.68 tons ha^{-1} increases in yield have also been reported (Peoples and Herridge, 1990). The measured benefits are dependent on the antecedent crop used as the reference criterion (Chalk, 1998).

Relative increases in N yield were in general higher than the relative increases in grain yield, suggesting the factors other than N were limiting grain yield. Many studies have verified that N is a major factor benefiting cereals following legumes compared with cereals following nonlegumes (e.g., Rowland *et al.*, 1988; Evans *et al.*, 1991; Chalk *et al.*, 1993). However, the reported response in grain yield may not be due entirely to N (Chalk, 1998). Improvements in soil structure, the breaking of pest and disease cycles in monoculture, and phytotoxic and allellopathic effects of different crop residues have all been responsible for the yield responses reported.

B. FERTILIZER NITROGEN RESPONSES

1. Fertilizer Nitrogen Equivalence

Various experiments have examined the N benefits of legumes to a subsequent nonlegume crop by comparing the effects of a number of rates of N fertilizer on the same crop grown after a nonlegume calculated as N fertilizer equivalence (NFE) (Clegg, 1982).

Estimates of the NFE of lucerne (*Medicago sativa* L.) to the following corn crop have been as high as 180 kg N ha^{-1} (Baldock and Musgrave, 1980; Voss and Shrader, 1984). The range of NFE values, as summarized in Table IX, is generally between 15 and 148 kg N ha^{-1} $year^{-1}$. Holford and Crocker (1997) reported that values of NFE for clover, lucerne, and medic for a first wheat crop varied from

Table IX

Nitrogen Fertilizer Equivalence (NFE) of Legumes as Reported by Different Workers

Legume/nonlegume	Yield response due to legume (kg ha^{-1})	NFE (kg N ha^{-1})	Reference
T. subterraneum/wheat	84–283	≤66	Brandt *et al.* (1989)
Lupinus albus L./wheat	29–83	22–182	Evans and Herridge (1987)
Vigna sinensis/wheat	30 (11)[a]	38 (13)	Bandopadhyay and De (1986)
Phaseouls aureus/wheat	49 (15)	68 (16)	Bandopadhyay and De (1986)
Arachis spps./wheat	23 (10)	28 (12)	Bandopadhyay and De (1986)
T.subterraneum/wheat	—	35–>120	Holford and Crocker (1997)
Medicago sativa/wheat	—	25–125	Holford and Crocker (1997)
M. scutellata/wheat	—	20–70	Holford and Crocker (1997)
Cicer arietinum/wheat	—	15–65	Holford and Crocker (1997)
M. sativa/maize	—	180	Voss and Shrader (1984)
M. sativa/maize	—	62	Fox and Piekielek (1988)
Vicia spps./maize–sorghum	—	65–135	Blevins *et al.* (1990)
Arachis spps./maize	39	60	Dakora *et al.* (1987)
V. sinensis/maize	95	30	Dakora *et al.* (1987)
Cajanus cajan/maize	57	38–49	Kumar Rao *et al.* (1983)
Sesbania rostrata/rice	—	50	Planiappan and Srinivasulu (1990)
S.aculeata/potato	100	48	Sharma and Sharma (1988)
Vigna radiata/potatoes	600	44	Sharma and Sharma (1988)
Green manures/rice[b]	—	34–148	Singh *et al.* (1991)

[a] Value in parentheses refers to companion crop.

[b] From more than 10 legume green manures in 24 experiments throughout the world.

70 to >120 kg ha^{-1} in the first year, 25–125 kg ha^{-1} in the second year, and 20–75 kg ha^{-1} in the third year, whereas corresponding values for chickpea were 35 kg in the first year, none in the second year, and 20 kg in the third year. Thus, the determination of NFE is resource intensive and provides an economic assessment of the value of including rotation in terms of fertilizer N saved (Chalk, 1998).

C. RESPONSES OF CEREALS TO THE ANTECEDENT LEGUME

1. Legume Nitrogen-Sparing Effects

The difference between the uptake of soil N or soil + fertilizer N by nonlegume and legume crops sown in adjacent plots is an estimate of N sparing, provided that net N mineralization under the different crops is equal (Chalk, 1998). Estimates of N-sparing reported ranged between 27 and 136 kg N ha^{-1}, depending on the stage of crop growth and reference crop (Chalk *et al.,* 1993). Evans *et al.* (1991) found that less soil N was taken up by lupin than adjacent cereal crops, whereas Herridge *et al.* (1995) measured a higher N-sparing effect in chickpea than wheat. Differences in amounts of mineral N in the soil profile also provide direct estimates of legume N sparing (Peoples *et al.,* 1995), and values ranging from 5 to 60 kg N ha^{-1}, depending on the crops selected, have been reported (Strong *et al.,* 1986a). However, additional mineral N under legume crops compared with cereals may result from the rapid mineralization of organic N derived from rhizodeposition rather than from a lower N demand by leguminous crops (Unkovich *et al.,* 1997). Thus, the concept of spared N needs to be reconsidered.

2. Amounts of Biological Nitrogen Fixation

It is important to have information on the actual N_2 fixation in the wide variety of environments (both spatial and temporal) and potential N_2 fixation in ideal field situations, as only then can efforts be made to manipulate the management practices to enhance N_2 fixation near to potential N_2 fixation (PNF).

The range of experimentally obtained values of N_2 fixation by temperate and tropical grain legumes of 0–97% N derived from the atmosphere (Ndfa) [0–450 kg N ha^{-1} (Evans *et al.,* 1989; Hardarson *et al.,* 1993; Herridge *et al.,* 1993; Peoples *et al.,* 1994a,b; Ladha *et al.,* 1996)] and from pasture legumes of 30–92% Ndfa [15–300 kg N ha^{-1} (Giller and Wilson, 1991; Ledgard and Steele, 1992; Peoples and Craswell, 1992; Thomas, 1992; Gault *et al.,* 1995)] reflects the inherent capacities of legumes to fix and accumulate N, the environmental constraints on those capacities, and the effects of cultural practices, experimental treatments, or both. Where soil fertility is high, legumes in the field thrive without fixing atmospheric N_2; under such conditions they may derive all their N requirements from soil N (Goh *et al.,* 1996) due to the altered source-sink relationship where an soil N apparently substituted for BNF (McNeill *et al.,* 1996). However, in the majority of soils, levels of plant available N are usually insufficient to fully satisfy the requirement of legume for N and the demand will be met by BNF.

For white clover in clover-based pastures, the amount of N_2 fixed may range from nil to more than 500 kg N ha^{-1} $year^{-1}$ in New Zealand. Where conditions are particularly favorable for white clover, a maximum of 670 kg N ha^{-1} $year^{-1}$ was reported during the establishment of a grass–white clover pasture on a subsoil supplied with additional P and K (Sears *et al.,* 1965). However, the usual range for grass–clover swards in New Zealand is between 100 and 350 kg N ha^{-1} (Caradus, 1990; Hoglund *et al.,* 1979; Ledgard *et al.,* 1990), with less than 100 kg N ha^{-1} in unimproved pastures and in dry areas (Ledgard *et al.,* 1990). High rates of fixation by white clover have also been reported from The Netherlands, with up to 565 kg N ha^{-1} $year^{-1}$ on a previously arable field on a clay soil (Mannetje, 1994). In Britain, the amount of N_2 fixed in a mixed grass–clover pasture can vary from nil to about 400 kg N ha^{-1} $year^{-1}$ and productive swards fix between 100 and 200 kg

N ha^{-1} year^{-1} (Reid, 1970; Munro and Davies, 1974; Laidlaw, 1988; Evans *et al.*, 1990). In Switzerland, between 270 and 370 kg N ha^{-1} were reported as being fixed by ryegrass–white clover swards (Böller and Nösberger, 1988). Major factors contributing to high BNF are high potential growth and a high percentage of N derived from the atmosphere (%Ndfa). This last parameter generally appears to be least affected by environmental conditions than total N_2 fixed (Danso *et al.*, 1992).

There has to be an upper limit on BNF (Dommergues and Steppler, 1987). Herridge and Bergersen (1988) postulated a theoretical upper limit of 635 kg N ha^{-1} for soybean (*Glycine max*) and more than 300 kg N ha^{-1} for pigeon pea (*Cajanus cajan*) and groundnut (*Arachis hypogea*). Although values approaching the theoretical limits may be achieved under optimal conditions (i.e., high legume yield and low soil nitrate) in practice, levels of N_2 fixation in fields of farmers may often be only a fraction of the potential fixation (Peoples *et al.*, 1995) as several ecological constraints may limit N_2 fixation by legumes. The identification of these constraints and the processes involved that limit N_2 fixation are important for the improvement of both agronomic practices and crop-breeding strategies.

3. Transfer of Biologically Fixed Nitrogen

The amount of legume N transferred to a subsequent cereal crop clearly depends on the amount of fixed N incorporated in a farming system. In crop rotations, the net N benefit from grain legumes may be small. For example, in a range of experiments with grain legumes varying in productivity and N_2 fixation, the net contribution from fixed N averaged only 15 to 20 kg N ha^{-1} and was frequently negative (Table X). This negative contribution occurred because the fixed N remaining in residue was less than soil-derived N removed in the harvested grains.

In contrast to grain legume cropping systems, where most of the legume N is removed in harvested grains, the contribution of fixed N can be substantial in legume-based pastures (Table XI). This is especially important in mixed cropping rotation (pasture/arable) systems of farming where N_2 fixation by the pasture phase is utilized to grow subsequent arable crops (Haynes and Francis, 1990; Francis *et al.*, 1995).

D. Relative Contribution of Fixed and Nonfixed Nitrogen to Crop Nitrogen Responses

An almost equal contribution of fixed N and nonfixed N toward N benefit has been reported in several studies (Kumar Rao *et al.*, 1987; Danso and Papastlianou, 1992; Chalk *et al.*, 1993). However, Senaratne and Hardarson (1988) found that the N benefit was predominantly due to nonfixed N when only roots were returned and the benefit declined further when both stover + roots were returned. Nevertheless, in general, a greater benefit is expected from the return of more leguminous residues (Chalk, 1998).

Table X

Estimates of Amounts (kg N ha^{-1}) of Nitrogen-Fixed, N Removal in Harvested Grains and Net Contribution of Fixed N to Subsequent Crop by a Range of Grain Legumes

Legume	Total plant N	Total fixed N	Total N removal in grain	Net contribution from fixed N	Reference
Chickpea	50–135	24–84	42–74	−18 to +10	Rennie and Dubetz (1986)
	60	6	46	−40	Smith *et al.* (1987)
	83–120	44–88	23–55	−7 to +65	Evans *et al.* (1989)
	139	104	94	+10	Armstrong *et al.* (1997)
Pea	61–91	28–177	27–91	−32 to +96	Evans *et al.* (1989)
	69–104	54–85	6–22	+52 to +73	Evans *et al.* (1997)
	116–140	53–86	49–91	−28 to +27	Smith *et al.* (1987)
	124–160	23–28	92–116	−64 to −93	Haynes *et al.* (1993)
	126–326	90–206	82–237	−34 to +17	Armstrong *et al.* (1994)
	212–238	160–196	154–170	+3 to +30	Rennie and Dubetz (1986)
	220	133	148	−15	Armstrong *et al.* (1997)
	220–227	133–183	135–162	−2 to +21	Peoples *et al.* (1995)
Lupin	48–131	18–80	5–78	−19 to +14	Smith *et al.* (1987)
	53–297	30–288	8–153	−41 to +135	Evans *et al.* (1989)
	292–347	111–126	220–278	−109 to −152	Haynes *et al.* (1993)
	303–420	249–317	154–266	+51 to +95	Armstrong *et al.* (1997)
Lentil	133–139	32	90–106	−58 to −74	Haynes *et al.* (1993)
	135–158	57–111	30–57	0 to +70	Smith *et al.* (1987)
	178–237	129–192	107–333	−143 to +26	Rennie and Dubetz (1986)
Faba bean	83	53	48	+5	Evans *et al.* (1989)
	107–161	64–115	18–84	−20 to +57	Smith *et al.* (1987)
	154–277	113–252	117–220	−8 to +40	Rennie and Dubetz (1986)
Cowpea	66–101	12–22	50–71	−57 to −18	Sisworo *et al.* (1990)
	130–200	66–117	82–85	−16 to +32	Awonaike *et al.* (1990)
Greengram	111–177	71–112	52–89	−19 to +23	Chapman and Myers (1987)
Soybean	78–108	26–33	47–64	−30 to −21	Sisworo *et al.* (1990)
	100–378	13–287	63–257	−116 to +67	Herridge and Bergersen (1988)
	329–402	290–312	262–296	−6 to +50	Awonaike *et al.* (1990)
	363–417	143–244	187–205	−44 to +39	Bergersen *et al.* (1985)
Field bean	173	48	98	−50	Haynes *et al.* (1993)

E. Role of Legume in the Gain or Drain of Soil Nitrogen

There is a great deal of variation in the literature concerning the potential of grain legumes in making a positive contribution to the N balance of cropping systems, given that a considerable amount of N is harvested and removed in grain. Eaglesham *et al.* (1982) presented a simple expression [Eq. (7)] to predict the net effect of grain legumes on soil N balance as

$$\text{N balance} = N_{fix} - N_{harv}, \tag{7}$$

where N_{fix} is N_2 fixed by the legume and N_{harv} is N removed in grain or other harvested portions.

McDonald (1989) expressed the concept in the form of Eq. (8) as

$$\begin{aligned}\text{N balance} &= (N_{leg} \times \text{Ndfa}) - (N_{leg} \times \text{NHI}) \\ &= N_{leg}\,(\text{Ndfa} - \text{NHI}),\end{aligned} \tag{8}$$

where $N_{fix} = N_{leg} \times \text{Ndfa}$ and $N_{harv} = N_{leg}$ x NHI, N_{leg} = legume N, Ndfa = proportion of N from BNF, NHI = proportion of crop N harvested in grain.

It is obvious from Eq. (8) that the N balance will be positive as long as NHI < Ndfa and it will be negative if NHI > Ndfa (Haynes *et al.,* 1993). In general, positive N balances are associated with the return of a greater amount of fixed N in crop residues compared with the removal of soil N in grain (Bergersen *et al.,* 1985; McDonagh *et al.,* 1993). Thus, grain legumes with high biomass N, low NHI, and high BNF have the greatest potential to contribute positively to the soil N pool (Chalk, 1998).

Some examples of both positive and negative estimates of N balances for grain legumes are shown in Table X. These N balances should be viewed with caution because of the overestimation of NHI as discussed previously. For example, Russell and Fillery (1996) found the NHI for lupin decreased from 0.78 to 0.56 when belowground biomass N (determined using foliar labeling) was included with aboveground N to give total plant N. In comparison to grain legumes, pastures are credited with supplying large amounts of fixed N to the system (Table XI). Chalk

Table XI

Amounts of Nitrogen (N_2) Fixation by The Legume Component of Different Legume-Based Mixed Pastures[a]

Legume component	N_2 fixation (kg N ha^{-1} $year^{-1}$)	Country
Trifolium repens L.	42–200	Uruguay
	83–283	Switzerland
	155	United Kingdom
	224–291	New Zealand
T. pratense L.	49–373	Switzerland
	49–277	Uruguay
Medicago littorallis L.	93–258	Canada
	51–172	Canada
	114–282	Austria
Lotus corniculatus	70–223	Uruguay
T. vesiculosum	20–60	United States

[a] From Ledgard and Giller (1995).

(1998) cautioned using single-season data to predict the long-term trends because both NHI and Ndfa show considerable spatial and temporal variations.

VII. CONCLUSIONS

This review reveals that crop residues of common cultivated crops are an important resource, not only as a source of significant amounts of nutrients for crops and hence agricultural productivity, but also affecting soil physical, chemical, and biological functions and properties and water and air quality. In general, major factors controlling residue decomposition are (i) factors related to the medium (soil) to which the residues are added, (ii) properties and quality of crop residues, (iii) residue management practices, (iv) climate, and (v) other factors. The development of an effective crop residue management system depends on a thorough understanding of these factors. It is difficult to predict decomposition from an individual property of organic residues, but when combined, these properties could accurately predict relative rates of decomposition from a broad range of important residues. Various indices, such as residue decomposability indices, resistance indices, and plant residue quality indices, have been developed, and although general trends have been observed, no unique relationship has been developed. This is partly due to different methodologies and approaches used by different workers to quantify the relationships. There is thus a need for standardizing methods for determining residue characteristics (e.g., lignin, polyphenol) and decomposition rates before a universal plant residue quality index could be developed and used to predict the nutrient release from decomposing residues.

Theoretical and predictive models have been developed to describe the decomposition process mathematically. An important conclusion from these models is the realization that in order to predict total mass loss, as well as the decomposition of primary litter fractions, the formation and turnover of secondary decay products need to be included. Although OM turnover models incorporating submodels or pools representing secondary decomposition products (e.g., microbial biomass) have been developed and combined with soil water and plant growth submodels to simulate cropping system dynamics, these models have not been tested extensively in terms of decomposition and nutrient release from different crop residues and management practices.

Crop residue management and tillage both affect the environment through its influence on losses of plant nutrients and chemicals to the environment causing pollution which is of great public concern. Awareness regarding the progressive degradation of soils has led to the search for a reliable measure of soil quality. Crop residue management is known to affect most of these soil quality indicators either directly or indirectly. It is perceived that soil quality is improved by the adoption of proper crop residue management practices. In addition, crop residues and their management also influence biological N_2 fixation by altering the inorganic N concentration in soil and hence their phytotoxicity. Considerable evidence has accumulated that the use of legumes can increase yields significantly. Indirect estimates indicate that leguminous crop residues can supply 15–148 kg chemical fertilizer equivalent N per hectare. Pasture legumes in mixed cropping systems can supply even higher amounts. A proper choice of legumes and their better management show a considerable potential in meeting part of the N demands for succeeding crops.

Comparisons of N recoveries from crop residue N and inorganic N fertilizers have shown that, in general, N recoveries from leguminous and nonleguminous residues are about one-half and one-eighth, respectively, of that from various forms of N fertilizers. Also, more legume N than fertilizer N is retained in soil and enters the organic N pool, whereas losses of legume N and fertilizer N are generally similar. Thus, there is a need to minimize losses of N from both systems by devising proper management practices for all cropping systems so that N mineralization synchronizes with crop N demand.

Several options are available to farmers in the management of crop residues. Ideally, crop residue management practices should be selected to enhance crop yields with minimum adverse effects on the environment. In the last two to three decades, several workers have examined the effect of residue management practices on the harvested yield of the following crop. Results from these experiments are conflicting because of a number of factors involved associated with residue quality, management and edaphic factors, health of the previous crop, and their complex interactions with various management factors in determining the ultimate crop yield. This indicates that no one residue management system is superior under all conditions. To overcome this problem, it is suggested that the effects of various constraints on crop production under different environments in each cropping system be identified and conceptualized to guide toward the best option. Multidisciplinary and integrated efforts by soil scientists, agronomists, ecologists, environmentalists, and economists are needed to design a system approach for the best choice of crop residue management practices for enhancing agricultural productivity and sustainability.

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