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Carbon and nitrogen turnover in adjacent grassland and cropland ecosystems

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Abstract. The effects of cultivation and soil texture on net and gross N mineralization, CO_2 evolution and C and N turnover were investigated using paired grassland and cropped sites on soils of three textures. Gross N mineralization and immobilization were measured using ^{15}N-isotope dilution. Grassland soils had high CO_2 evolution and gross N mineralization rates, and low net N mineralization rates. Cropland soils had low CO_2 evolution rates but had high net and gross N mineralization rates. Grassland soils thus had high immobilization rates and cropland soils had low immobilization rates. Cultivation increased N turnover but reduced C turnover. The data suggest that the microflora in grassland soils are N limited, while those of cropland soils are limited by C availability. Increasing clay content reduced N turnover. C turnover was less clearly related to texture. Differences in the immobilization potential of substrates help explain why agricultural soils have higher N losses than do grassland soils.

Introduction

Although general principles governing carbon-nitrogen interactions during decomposition are beginning to be understood, problems of characterizing soil organic matter quality and belowground inputs of carbon have hampered efforts to study C–N interactions in the soil (McClaugherty et al., 1982, Paul 1984). Turnover of soil organic N has also proved very difficult to study because it can only be measured using tracer techniques (Nishio et al., 1985, Vitousek and Matson 1985, Paul and Juma 1981, Jansson 1958). Soil N turnover is apparently regulated by interactions with soil physical properties; turnover appears to be inhibited by clay (Ladd et al., 1981, Campbell and Souster 1982, Van Veen et al., 1984, Schimel et al., 1985a, Schimel et al., 1985b).

Conversion of grasslands to croplands affects carbon fluxes by reducing carbon inputs to the soil, reducing root:shoot ratios in dominant vegetation and increasing decomposition rates (Anderson and Coleman 1985, Coleman et al., 1984, Voroney et al., 1981). Conversion may also alter soil texture by increasing erosion rates (Ruhe and Walker 1968). Paired cultivated and native ecosystems can provide an experimental setting for

studies on the effects of carbon inputs and soil texture on soil carbon and nitrogen turnover. Studies on such paired sites in North Dakota showed that that ratio of CO_2 evolution to net N mineralization was higher in grassland than cropland soils (Schimel et al., 1985b). We hypothesized that substrate quality is different between grassland and cropland soils, which results in grassland soils having more immobilization per unit gross mineralization than cropland soils. The objective of this study was to test this hypothesis by measuring C and N mineralization and indices of C and N turnover as affected by cultivation and soil texture.

Methods

Site description

Soils from the summit positions of three paired grassland and cultivated toposequences were used in this study. The sites, located in southwestern North Dakota, have been the subject of intensive studies of erosion and organic matter changes resulting from cultivation (Aguilar 1984, Kelly 1984, Schimel et al., 1985b, Schimel et al., 1985c). The cultivated sites had all been in wheat-fallow rotations for 45 y at the time of sampling. The toposequences were on three different parent materials with varying textures (sandstone, siltstone, and shale), with grassland and cropland components separated by fences or, in one case, by a road. The siltstone site has never been fertilized; the other sites have received about $10\,kg\,N \cdot ha^{-1} \cdot y^{-1}$. The soils of all three sites were classified as typic Haploborolls (Soil Survey Staff 1975). Detailed site and soil descriptions are in Schimel et al., (1985b) and Aguilar (1984). Selected soil properties are shown in Table 1. Organic C and N are shown in Table 5.

Field sampling

Soil samples for incubations were collected in August 1984, at which time the cropland sites had been recently harvested. Three 7.5-cm-diameter by 10-cm-depth cores were composited for each sample and three samples were collected for each site. The depth increment was within the surface horizon of every soil sampled. The samples were kept in a cooler with ice and returned to Fort Collins for analysis.

Incubations

Soil samples were mixed gently and all obvious organic debris, including live and dead roots, was removed by hand to minimize disturbance of soil structure. Moisture contents were determined by drying subsamples at 105°C for 48 h. Initial NH_4^+ and NO_3^- contents were determined by extraction of 3 subsamples with $2\,mol \cdot L^{-1}$ KCl and subsequent colorimetric analysis. Each field sample was then split into twelve 50-g dry weight aliquots for incubation. Nine aliquots were used for aerobic in-

Table 1. Selected soil properties: clay content and percent H_2O at -0.03 MPa

Parent Material	Treatment	% clay	% H_2O at -0.03 MPa (volumetric)
Sandstone	Grass	13	20
	Crop	4	19
Shale	Grass	38	29
	Crop	26	32
Siltstone	Grass	24	38
	Crop	36	43

cubations, and the remaining three were used for determination of chloroform-labile C and N [microbial biomass C and N of Jenkinson and Powlson (1976)]. At day 0, each aliquot was brought to field capacity less 2.5 ml · g^{-1}. Two and a half milliliters of 100 μg N · ml^{-1} solution containing 99.2 atom percent ^{15}N-$(NH_4)_2SO_4$ as the N sources were then added, bringing the moisture content up to field capacity, and adding 250 μg of ^{15}N as a tracer. Moisture contents used are shown in Table 1. The ^{15}N solution was carefully mixed into the soil with a glass rod to give as uniform a distribution of label in the soil as possible.

The samples were incubated in sealed jars with CO_2 traps as described in Schimel et al., (1985b). Sufficient headspace was allowed to maintain aerobic conditions. Three aliquots from each sample were destructively sampled at days 4, 12, and 30 and analyzed for NH_4^+, $^{15}NH_4^+$, NO_3^-, and $^{15}NO_3^-$. CO_2 evolution was also determined at each of those times. Chloroform labile C, N, and ^{15}N were determined at day 12 and organic N and ^{15}N were determined at day 30.

Chemical and isotope analyses

Organic carbon was measured on each sample after digestion with dichromate in concentrated H_2SO_4-H_3PO_4 (Snyder and Trofymow 1984). Organic N was determined following a micro-Kjeldahl procedure, using a block digestor. Inorganic N (NH_4^+ and NO_3^-) was determined using Technicon I Autoanalyzer (Technicon Industrial Systems, Tarrytown, New York). Carbon dioxide was determined in 1 mol · L^{-1} NaOH traps by titrating excess base with 1 mol · L^{-1} HCl in the presence of $BaCl_2$.

Chloroform-labile C and N were determined using the procedure of Jenkinson and Powlson (1976), with modifications described in Voroney (1983) and Voroney and Paul (1984). Chloroform-labile ^{15}N was determined by measuring the ^{15}N content of the N flush and dividing that by the same K_n used for the unlabelled N (Voroney 1983).

^{15}Nitrogen was determined on KCl extracts and digests by diffusion of NH_3^0 into 1 mol · L^{-1} HCl (Adamsen and Reeder 1984). $^{15}NO_3^-$ was converted to NH_3^0 using Devarda's alloy. Samples were treated with Devarda's

alloy for 18 h at 80°C in 100-ml diffusion tubes. Rubber stoppers with narrow glass tubes were placed in the diffusion tubes to allow escape of H_2 gas. The tubes were then sealed and the temperature increased to 105°C for 48 h. Twenty to 100 μg of N are required for analysis by mass spectrometer. Because of the low concentrations of NH_4^+ in many of our samples, and the restricted volume of the diffusion tubes, some samples had to be repeatedly diffused. In these cases, the acid trap was transferred from tube to tube, until sufficient N had been trapped. Following all diffusions, the acid traps were dried and sent to Isotope Services, Inc., Los Alamos, New Mexico, for mass spectrometric analysis. All mass spectrometric analyses were performed in duplicate.

Isotope calculations

Estimates of N fluxes were obtained by either mass balance or isotope dilution calculations. Mass balances were computed by multiplying % atom excess (^{15}N sample-^{15}N background) times pool size and calculating change over time. ^{15}N:^{14}N ratios can then be used to calculate fluxes of unlabelled N (Clark 1977, Schimel et al., 1986). Net N mineralization was calculated from $\Delta(NH_4^+ + NO_3^-)$ for each time interval. Gross N mineralization was calculated using equations from Kirkham and Bartholomew (1954) and Jansson (1958). These equations assume that flux of ^{15}N from microbes or organic matter to inorganic N (N_i) is negligible (i.e., no remineralization of added label). This assumption was probably not met after the first 4 d of incubation. The equation used was

$$\text{Gross mineralization} = \text{(Net mineralization) [Log (initial } {}^{15}N_i \cdot \text{Final } {}^{14+15}N_i / \text{initial } {}^{14+15}N_i \cdot \text{Final } {}^{15}N_i)]/\text{Log (Final } {}^{14+15}N_i/\text{Initial } {}^{14+15}N_i) \quad (1)$$

The final units are $\mu g\, N \cdot g^{-1} \cdot d^{-1}$. This equation is derived in Kirkham and Bartholomew (1954) and is only valid when mineralization exceeds immobilization; i.e., net mineralization is a positive quantity. Other equations are applicable if net immobilization occurs or if mineralization and immobilization are equal (Kirkham and Bartholomew 1954, 1955; Shen et al., 1984).

All statistical analyses were performed using subprograms ANOVA and REGRESSION in SPSS-PC on an IBM PC-XT. Significance was assumed at $P \leqq 0.001$ unless otherwise stated.

Results and discussion

Mineralization and immobilization

NH_4^+ dropped to nearly undetectable levels by day 4 in all soils and remained so for the remainder of the experiment, indicating high rates of

Table 2. Net and gross mineralization during the first 4 days of incubation and CO_2 evolution from grassland and cropland soils on three parent materials. C and N turnover are shown in % organic C or N mineralized per day during days 1–4 of the incubation

Parent Material	Treatment	Net N mineralization ($\mu gN \cdot g^{-1} \cdot d^{-1}$)	Gross N mineralization ($\mu gN \cdot g^{-1} \cdot d^{-1}$)	CO_2 evolution ($\mu gC \cdot g^{-1} \cdot d^{-1}$)	N turnover gross min./organic N ($\% \cdot d^{-1}$)	C turnover CO_2/organic C ($\% \cdot d^{-1}$)
Sandstone	Grass	1.4 ± 0.9*	5.5 ± 0.5	43 ± 1.7	0.33 ± 0.03	0.36 ± 0.01
	Crop	2.6 ± 0.7	6.4 ± 0.8	23 ± 5.4	0.48 ± 0.06	0.22 ± 0.05
Shale	Grass	1.2 ± 0.9	6.4 ± 0.1	64 ± 1.1	0.21 ± 0.00	0.30 ± 0.00
	Crop	3.3 ± 0.4	6.0 ± 0.1	39 ± 3.7	0.32 ± 0.07	0.26 ± 0.02
Siltstone	Grass	1.2 ± 0.0	6.7 ± 0.2	62 ± 1.3	0.24 ± 0.01	0.21 ± 0.00
	Crop	2.9 ± 0.6	6.5 ± 1.4	27 ± 8.5	0.38 ± 0.08	0.17 ± 0.05

*± 1 SE.

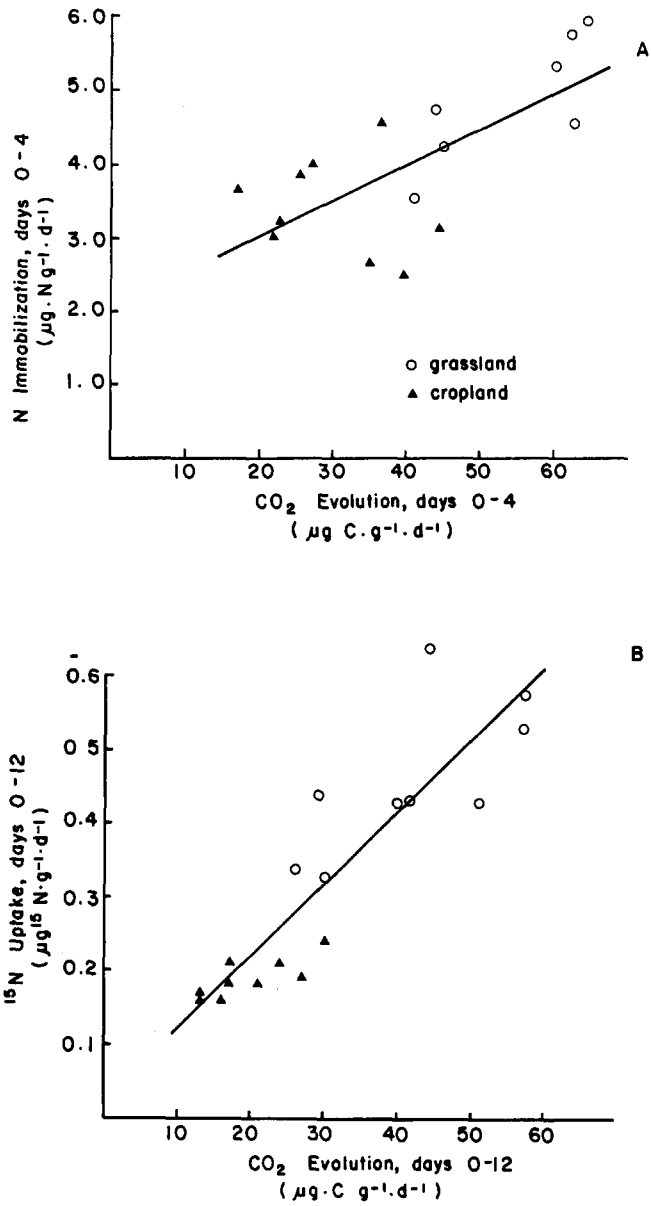

Figure 1. (a) N immobilization increased with increasing CO_2 evolution (respiration). Grassland soils had high CO_2 evolution and immobilization rates; cropland soils had the opposite. The equation is $y = 2.1 + 0.05 * x$. Both the slope and intercept were significantly different from O ($P = 0.002$). (b) ^{15}N uptake into $CHCl_3$-labile nitrogen also increased with increasing CO_2 evolution. The same clustering of grassland and cropland N uptake and respiration rates was seen as for Fig. 1a, above. ^{14}N immobilization can not be calculated from this data because of uncertainties in the ^{15}N:^{14}N ratio of the source pool.

Table 3. ^{15}N budget for the incubation experiment. $^{14}NH_4^+$ and $^{15}NH_4^+$ were virtually undetectable throughout the experiment. Average recovery of ^{15}N in micro-Kjeldahl digests at day 30 was 100% for all soils, showing that gaseous losses were not significant.

Parent Material	Treatment	Day of incubation				
		0	4	12		30
		$^{15}NH_4^+$-N$^+$	$^{15}NO_3^-$-N*$^+$	$^{15}NO_3^-$-N*$^+$	$CHCl_3$-Labile $^{15}N^+$	$^{15}NO_3^-$-N*$^+$
Sandstone	Grass	10	3.1 ± 1.0#	3.0 ± 0.7	4.5 ± 1.0	2.5 ± 0.1
	Crop	10	6.0 ± 0.7	6.4 ± 0.5	2.1 ± 0.7	3.8 ± 2.2
Shale	Grass	10	1.9 ± 0.6	1.6 ± 1.2	6.6 ± 1.0	2.0 ± 0.5
	Crop	10	5.9 ± 0.6	6.3 ± 0.8	2.7 ± 0.1	1.7 ± 2.9
Siltstone	Grass	10	1.6 ± 1.2	1.9 ± 1.1	6.5 ± 2.3	2.5 ± 0.9
	Crop	10	6.2 ± 0.2	6.4 ± 0.1	2.1 ± 0.2	4.6 ± 1.3

*After correction for background.
+Units are $\mu g\ ^{15}N \cdot g^{-1}$.
± 1 SE.

flux through that pool. During the 0 to 4 day period the grassland soils had significantly lower net mineralization rates but significantly higher CO_2 evolution rates than did the cropland soils (Table 2). Gross N mineralization rates were not affected by cultivation; thus, the grassland soils had higher immobilization rates than the cropland soils. A significant relationship between immobilization (computed as gross-net N mineralization) and respiration was found, with grassland soils having high immobilization and respiration rates and cropland soils having the opposite (Fig. 1a). The rate of ^{15}N uptake into $CHCl_3$-labile N was also highly correlated with CO_2 evolution ($R^2 = 0.79$; $p < 10^{-4}$) (Fig. 1b). The difference in immobilization rates resulted in much more inorganic ^{15}N being recovered in the cropland than the grassland soils at day 4 (Table 3). Most of the immobilized N remained in the $CHCl_3$-labile fraction at day 12, and so the disappearance of inorganic ^{15}N was probably due to microbial uptake. The amount of ^{15}N recovered in the $CHCl_3$-fraction was higher in the grassland than the cropland soils. The % atom excess was also significantly higher, showing that uptake of ^{15}N per unit $CHCl_3$-labile N was higher in the grassland than in the cropland soils.

Data from the 0 to 4 day portion of the experiment suggests that the limiting factor for microbial activity is different in grassland and cropland soils. Microbial populations in the grassland soils were strongly–limited. Despite their high respiration rates, net mineralization rates were low. Rates of immobilization were high, and rates of ^{15}N uptake per unit $CHCl_3$-labile N, which is related to microbial biomass (Voroney 1983), were higher in grassland than cropland soils. The microflora of fertilized and unfertilized cropland soils were limited by the availability or quality of carbon substrate. Despite low CO_2 evolution rates, they had high rates of net mineralization, low rates of immobilization, and lower uptake of tracer ^{15}N per unit $CHCl_3$-labile N than did the grassland soils. The difference in limiting factors between the grassland and cropland soils suggests a difference in the amount and quality of the organic substrates available to the microorganisms in the two types of ecosystem. While the grasslands are higher in total and microbial N, much of this N may be associated with resistant or protected substrates or organisms slowly decomposing such substrates. Respiration rates declined with time for all soils (Fig. 2b). Net N mineralization rates declined markedly with time in the cropped soils but increased in grassland soils, again suggesting that there were differences in substrate quality or amount (Fig. 2a). Further evidence for a change in substrate quality comes from the lack of correlation between respiration and gross N mineralization ($P > 0.5$).

Comparing the $^{15}NO_3^-$ pools at days 12 and 30 gives evidence for significant NO_3^- immobilization (Table 3), as the absolute amount of $^{15}NO_3^-$ declined markedly between day 10 and day 30 in several of the soils. A rate of NO_3^- immobilization can be calculated from the product

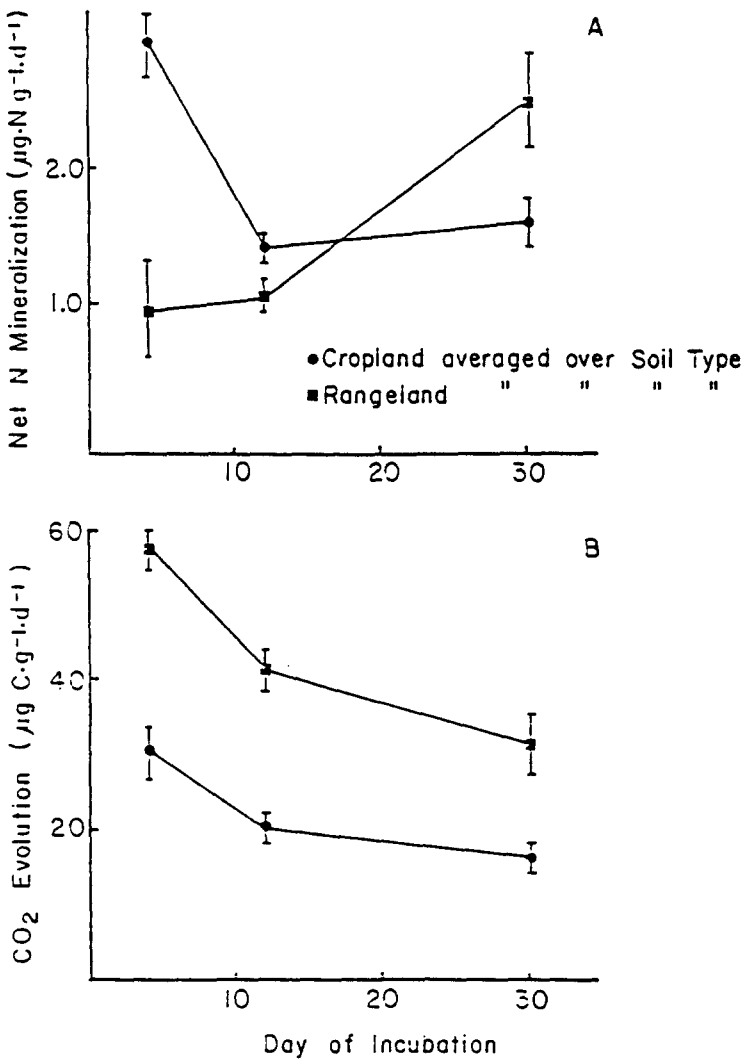

Figure 2. (a) Daily rates of net mineralization for the 0–4, 4–12, and 10–30 intervals. (b) Daily rates of CO_2 evolution for intervals as above. Differences between parent materials were small compared to differences due to cultivation and so data are shown averaged over parent materials, ± 1 SE.

of the decline in $^{15}NO_3^-$ and the mean ^{14}N:^{15}N ratio of the NO_3^- pool (Table 4). This rate may be an underestimate because of concurrent mineralization of organic ^{15}N. Rates of NO_3^- immobilization ranged from 0.3 to 2.6 $\mu g\, N \cdot g^{-1} \cdot d^{-1}$. This is in contrast to frequent reports that NO_3^- is not immobilized except in heavily carbon-amended soils (Jansson 1958, Shen et al. 1984, Nishio et al. 1985). During the same interval, the amount of $^{15}NO_3^-$ in two of the grassland soils increased, showing that more

Table 4. NO_3^- immobilization calculated from the decline in $^{15}NO_3$-mass and the linear mean enrichment of that pool during days 12–30

Parent material	Treatment	$^{15}NO_3^-$ decrement days 12–30† ($\mu g \cdot {}^{15}N \cdot g^{-1}$)	Mean ^{14}N:^{15}N ratio, days 12–30	NO_3^- immobilization ($\mu g \cdot N \cdot g^{-1} \cdot d^{-1}$)
Sandstone	Grass	0.5	11.9	0.3
	Crop	2.6	9.5	1.4
Shale	Grass	*	—	—
	Crop	4.6	10.1	2.6
Siltstone	Grass	*	—	—
	Crop	1.8	8.5	0.8

† From Table 3.
* $^{15}NO_3^-$ increased; see Table 3.

labelled N was being released than was being taken up. Gross mineralization cannot be calculated directly for the grassland soils because of uncertainty about the ^{15}N enrichment of the mineralizing pool.

C and N turnover

An index of N turnover (N_t) was calculated from gross-mineralization/organic N. C turnover (C_t) was calculated similarly from CO_2 evolution/organic C (Tables 2, 5). Both indices were calculated from the day 0–4 rates and are expressed as $\% \cdot d^{-1}$. Cultivation affected both C_t and N_t but in opposite ways (Table 2). Cultivation increased N_t significantly, but decreased C_t. The same pattern can be seen if turnover is expressed as mineralization over $CHCl_3$-labile organic matter, rather than total organic matter (Tables 2, 5). If N is limiting, one would expect the biota to be conservative of N and to maintain N in organic forms or within the biomasss, rather than releasing it as N_i (Vitousek 1982). If carbon quality is restrictive, that should slow down carbon turnover by reducing decompositon rates (Melillo et al., 1984). C turnover data from previous studies on this site showed the same effects of cultivation as presented here (Schimel et al., 1985b).

Table 5. Organic and $CHCl_3$-labile C and N in cropped and grassland soils on three parent materials. $CHCl_3$-labile C and N have been corrected using the K_C and K_N values of Voroney (1983) [$K_C = 0.41$, $K_N = 1.86$ (C flush/N flush)$^{-0.879}$, mean $K_N = 0.32 \pm 0.03$]

Parent material	Treatment	Organic C	Organic N	$CHCl_3$-labile C	$CHCl_3$-labile N
		μg C or N $\cdot g^{-1}$			
Sandstone	Grass	12,010 ± 400*	1676 ± 34	954 ± 141	164 ± 23
	Crop	10,360 ± 1,300	1316 ± 113	604 ± 129	104 ± 19
Shale	Grass	21,300 ± 1,900	3041 ± 90	1841 ± 58	316 ± 9
	Crop	15,000 ± 2,200	1873 ± 70	978 ± 129	168 ± 20
Siltstone	Grass	29,050 ± 3,400	2706 ± 45	1558 ± 143	267 ± 21
	Crop	15,750 ± 400	1681 ± 9	629 ± 158	110 ± 27

* ± 1SE.

Estimates of turnover in this study, based on gross N mineralization, show that N_t rates declined with increasing clay content in grassland soils. Under grassland conditions, C_t was highest in the lowest clay soil, but was next in the highest clay soil. No clear trend with texture in C_t was seen under cropped conditions.

Conclusions

The respiration and N-turnover data yield important information about C–N interactions during decompositon in soils. Soils with high rates of CO_2 evolution but low rates of net N mineralization had high rates of N immobilization. The high immobilization rates suggest that the high CO_2 evolution rates were associated with a growing microbial biomass. Soils with low rates of CO_2 evolution and high rates of net mineralization had low rates of immobilization, suggesting a steady-state or decaying biomass. Differences in respiration and N-immobilization rates between grasslands and croplands suggest that the microbiota are N limited in grasslands but C limited in croplands. This was true for both fertilized and unfertilized soils. This change in limiting factors may result from differences in substrate quality and abiotic controls over decomposition rate between grasslands and agricultural soils.

Through management systems which mimic grassland ecosystems by slowing decompositon or increasing immobilization, it may be possible to match N release synchronously to crop requirements (Woodmansee 1984). Developing such systems will require careful management of the soil microbiota as part of an overall management system (Coleman et al., 1984).

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