

Carbon and Nitrogen Mineralization Rates after Application of Organic Amendments to Soil

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ABSTRACT

The objective of this study was to quantify C and N mineralization rates from a range of organic amendments that differed in their total C and N contents and C quality, to gain a better understanding of their influence on the soil N cycle. A pelletized poultry manure (PP), two green waste-based composts (GWCa, GWCb), a straw-based compost (SBC), and a vermi-cast (VER) were incubated in a coarse-textured soil at 15°C for 142 d. The C quality of each amendment was determined by chemical analysis and by ¹³C nuclear magnetic resonance (NMR). Carbon dioxide (CO₂-C) evolution was determined using alkali traps. Gross N mineralization rates were calculated by ¹⁵N isotopic pool dilution. The CO₂-C evolution rates and gross N mineralization rates were generally higher in amended soils than in the control soil. With the exception of GWCb all amendments released inorganic N at concentrations that would be high enough to warrant a reduction in inorganic N fertilizer application rates. The amount of N released from PP was high indicating that application rates should be reduced, or alternative amendments used, to minimize leaching losses in regions where ground water quality is of concern. There was a highly significant relationship between CO₂-C evolution and gross N mineralization ($R^2 = 0.95$). Some of the chemically determined C quality parameters had significant relationships ($p < 0.05$) with both the cumulative amounts of C and N evolved. However, we found no significant relationships between ¹³C NMR spectral groupings, or their ratios, and either the CO₂-C evolved or gross N mineralized from the amendments.

THE APPLICATION of organic amendments to soil is increasing as both an environmentally favorable waste management strategy and as a means of improving soil organic matter (SOM) content in low-fertility soils. The incorporation of soil amendments requires appropriate management strategies that will support crop production and protect water quality (Mamo et al., 1999). Metropolitan Perth, Western Australia, has a population of approximately 1.8 million people and produces >1.2 million Mg of organic waste annually. A combination of waste reduction strategies, including the production of organic soil amendments, are currently being employed in an effort by the waste management industry to reduce pressure on landfill sites. Currently commercial organic amendments are produced from domestic garden waste (lawn and tree clippings) and/or manures from intensive commercial animal (poultry, swine, and bovine) production. Treatment of these waste products

includes composting for various lengths of time, processing through earthworms, or drying and pelletizing.

The horticultural industry is being targeted as the main end-user of organic amendments due to the high economic returns and close proximity of this industry to urban centers where composting facilities are established; thus minimizing transport costs. Western Australian vegetable production accounts for about 40% of Australia's vegetable exports and covers around 11 100 ha (Hutchison, 2002). This area is dominated by coarse-textured soils with low SOM and clay content (<2%), poor water and/or nutrient retention, and low microbial populations. With two or three crops (mainly carrots, lettuce, potatoes, or broccoli) produced per year and inorganic N fertilizer applications at rates up to 400 kg N ha⁻¹ per crop, the potential for N to be leached from these systems is high even when inorganic fertilizer application is split throughout the growing season (Hutchison, 2002). Traditionally raw animal manures have been applied to soil but the practice was legislated against due to problems with odor and insect breeding (especially stable fly) in unprocessed manure piles (Hutchison, 2002). Composting has proven to be an effective process for reducing fly and odor problems in manures (Cook et al., 1997).

Considerable research has shown the benefits of using composts and other amendments to improve soil physical (water holding capacity, porosity, and bulk density), chemical (pH, electrical conductivity, and nutrient content), and biological properties such as soil microbial populations and plant growth (Mamo et al., 1999; Stoffella and Kahn, 2001; Wolkowski, 2003). However, there are few detailed studies on how amendments impact on soil processes and thus effect crop production. Organic amendments vary greatly in their composition and degree of stabilization and thus their capacity to release nutrients is often ignored or overestimated (Sikora and Szmidt, 2001). Soil N supply from organic amendments depends on both the initial availability of inorganic N in the amendments and also the longer-term rate of mineralization and subsequent immobilization of N. This means that organic amendments can either be a source of plant available N or compete with plants for it. To successfully manage nutrient cycling from organic amendments it is therefore necessary to know their decomposition rate and the influence this will have on N processes within the soil (Ambus et al., 2002; Gabrielle et al., 2004).

Nitrogen mineralization-immobilization turnover de-

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Abbreviations: GWCa and GWCb, 18- and 24-week green waste-based composts, respectively; MB-C, microbial biomass carbon; MB-N microbial biomass nitrogen; NMR, nuclear magnetic resonance; PP, pelletized poultry manure; SBC, straw-based compost; SOM, soil organic matter; VER, vermi-cast.

depends on a variety of factors with the C to N ratio being regarded as the single most predictive measurement of the N mineralization capacity of composts (Sikora and Szmidt, 2001). The C to N ratio is commonly used to assess whether plant residues will release or immobilize inorganic N (Nicolardot et al., 2001; Hadas et al., 2004). In situations where plant growth occurs over several months or years the C to N ratio of compost can be an adequate predictor of the amendment's impact on N cycling and hence on crop performance. However, during shorter crops (e.g., 10 wk for a summer-grown lettuce crop) other predictive measures may be required (Gilmour, 1998). Carbon complexity influences the rate of microbial SOM decomposition (Bernal et al., 1998b; Kogel-Knaber, 2002) and thus regulates amendment stability and N release (Bernal et al., 1998a), as easily degradable forms of C are preferentially utilized by microbial populations. The total C to N ratio will not necessarily give a good indication of an amendment's potential N supply (Ambus et al., 2002). Here measures of the more bioavailable C components of the amendment may need to be defined, such as those determined by chemical characterization and solid state ^{13}C nuclear magnetic resonance (NMR). The size and activity of the soil microbial community will also influence N cycles within the soil (Bengtsson et al., 2003). Previous studies have identified ^{13}C NMR characteristics of organic matter and compost or suggested maturity indexes for composts based on chemical C characteristics, bioassays, and inorganic N traces (Bernal et al., 1998a, 1998b, 1998c; Rowell et al., 2001; Smernik and Oades 2000a). Rowell et al. (2001) used ^{13}C NMR parameters to model net N mineralization from biosolid-based amendments. However, there has been little work on these relationships to the absolute supply capacity of the organic amendment (i.e., gross N mineralization).

The aim of this study was to quantify the pattern of C and N decomposition (mineralization rates) from a range of organic amendments that differ in their C to N ratio and C quality. The size and activity of the soil microbial population was also measured to determine what, if any, differences were observed after incorporation of the amendments. This information is required to develop guidelines for amendment incorporation into commercial horticultural production systems.

MATERIALS AND METHODS

Soil

The 0- to 10-cm layer of a coarse-textured sand (0% silt, 2% clay, 98% sand; pH [1:5] = 5.85 in 0.5 M CaCl_2 ; electrical conductivity [1:5] = 4.25 mS m^{-1} ; total C = 0.72%; total N = 0.05%) was collected from the Medina Horticultural Research Station, near Perth, Western Australia (32°58.5' S, 115°52' E). Soil was sieved to 5 mm and slightly dried (to approximately 40% water holding capacity) at 25°C for 36 h. Soil was then mixed and stored for 5 d, in the dark at 4°C before use.

Amendments

Five commercially produced organic amendments with varying C qualities were assessed (Table 1). Compressed poultry

Table 1. Chemical characteristics of each amendment as determined by wet chemistry and ANKOM analysis ($n = 4$ for NH_4^+ and NO_3^- values $n = 2$ for other parameters).

Amendment†	Chemical properties					ANKOM analysis§							
	Total C	Total N	mg N kg ⁻¹			pH	EC‡	%					
			NH ₄ ⁺ -N	NO ₃ ⁻ -N	Ratio			ADF¶	Cellulose	Lignin	Ash		
	%					mS m ⁻¹	g						
PP	40.86 (0.06) e#	4.03 (0.00) e	69.62 (0.26) d	0.81 (0.11) a	10.1	5.40	5.73	0.48	38.00 (0.92) a	23.77 (0.7) d	13.02 (0.7) d	2.45 (0.1) a	
GWCa	22.32 (0.02) c	1.39 (0.02) b	2.08 (0.03) c	8.89 (0.14) b	16.1	7.28	4.15	0.17	38.68 (0.65) a	14.29 (0.2) b	8.08 (0.3) b	17.48 (0.3) c	
GWCB	29.97 (0.01) d	1.19 (0.01) a	0.42 (0.13) ab	0.03 (0.01) a	25.1	7.16	13.89	0.14	50.63 (1.58) b	21.71 (0.1) c	14.95 (0.3) e	15.22 (0.3) b	
SBC	16.24 (0.00) a	1.42 (0.00) c	0.48 (0.08) b	8.64 (0.70) b	11.4	6.79	2.69	0.17	37.34 (0.60) a	8.35 (0.8) a	5.10 (0.1) a	25.08 (0.1) e	
VER	20.91 (0.06) b	1.72 (0.00) d	0.17 (0.04) a	1.04 (0.07) a	12.2	7.51	2.66	0.21	39.78 (0.22) a	9.78 (0.3) a	8.89 (0.0) c	22.35 (0.1) d	

† PP, pelletized poultry manure; GWCa, 18-wk green waste-based compost; GWCB, 24-wk green waste-based compost; SBC, straw-based compost; VER, vermi-cast.

‡ Electrical conductivity.

§ Values for ANKOM analysis are percentages of the total amount of carbon in the amendment.

¶ Acid detergent fiber.

Numbers in parentheses are standard error of the mean. Values with the same letter are not significantly different based on 95% confidence intervals.

manure pellets (PP) (3×10 mm) comprising 90% manure and 10% straw were manufactured using manure that was 6 to 10 wk old. During the manufacturing process the manure was heated to approximately 70°C and dehydrated. Green waste-based composts containing 90% green waste (lawn and tree clippings) and 10% poultry manure were composted using 1.5- \times 20-m aerobic windrows for 18 wk (GWCa) or 24 wk (GWCb). GWCa is what the manufacturer terms "finished" and ready for sale. GWCb was derived from an earlier batch of GWCa that underwent a further maturation phase to simulate the storage period that typically occurs between the purchase and application of the finished product. It was necessary to generate GWCb from an earlier batch of GWCa so that both products were ready to use at the start of this incubation study. Straw-based compost (SBC) was composed of 35% straw, 30% pig manure, 5% sawdust, and 30% spent mushroom compost, based on pig manure and straw. This had been composted for 12 wk also using windrows and was classified as "finished." The vermi-cast (VER) amendment was the casting collected from earthworms fed a mixture of 20% grape marc (seed, skin, and stalks) and 80% semicomposted straw and manures (pig, poultry, and bovine). The castings contained small worm eggs, which were removed by hand to prevent hatching. All amendments were obtained from producers a maximum of 3 d before the start of the incubation. Amendments were sieved (<3 mm) and any large material was cut to <3 mm before storage in the dark at 4°C until use. All percentages reported for the compost mixes above are approximates on a per volume basis.

A subsample of each amendment was dried (40°C) and finely ground for characterization. Total C and N contents were determined using a CHN 1000 elemental analyzer (LECO, St. Joseph, MI). Acid detergent fiber, cellulose, and lignin contents were determined in duplicate using a fiber analyzer from ANKOM (Macedon, NY). Total ash content was determined by loss on ignition (3 h, 500°C).

Solid state ^{13}C cross polarization (CP) NMR spectra were acquired at a frequency of 50.3 MHz on a Varian (Palo Alto, CA) Unity 200 spectrometer. Samples were packed in a 7-mm-diameter cylindrical zirconia rotor with Kel-F end-caps and spun at 5000 ± 100 Hz in a Doty Scientific (Columbia, SC) Magnetic Angle Spinning probe. Cross polarization spectra were acquired using a contact time of 1 ms. The recycle delay was set to at least 20 times the "null time" of the inversion recovery experiment resulting in a 0.5-s delay for the composts (GWCa, GWCb, and SBC) and a 2-s delay for VER and PP. Spin counts were performed using the method of Smernik and Oades (2000a, 2000b) using glycine (AR grade; Ajax Chemicals, Seven Hills, NSW, Australia) as a reference. A correction was made for signal loss through $T_{1\rho}\text{H}$ relaxation during the contact time of the CP experiment. The average $T_{1\rho}\text{H}$ was determined by linear regression of \ln (intensity) against spin lock time for the variable spin lock experiments (Smernik et al., 2002). The observed C values were highly correlated with C content ($R^2 = 0.88$), so although 30 to 56% of the potential ^{13}C NMR signal was unaccounted for, much of the signal loss was likely to be nonselective and should not change the distribution of signal through the spectrum (Smernik and Oades, 2000b).

Laboratory Incubations

Each organic amendment was uniformly incorporated into subsamples of soil at a rate equivalent to $30 \text{ m}^3 \text{ ha}^{-1}$ (12 g dry material per 500 g soil) which is the recommended field application rate for green waste-based composts. Deionized water was used to bring all soils to 75% water holding capacity

(9% gravimetric moisture) and this was adjusted weekly. Each subsample of soil was stored in a plastic container (750 mL) with lids containing 3-mm-diameter holes to enable gas exchange. Containers were randomized and incubated in the dark for up to 142 d (approximately one winter crop cycle for lettuce production) at 15°C (average daily temperature during winter). Containers were destructively sampled on Days 3, 9, 16, 23, 37, 51, 82, 107, or 142.

Carbon Dioxide Evolution

Soil $\text{CO}_2\text{-C}$ evolution rates were determined using static incubation vessels (7 d, 15°C) containing the equivalent to 50 g dry soil and 10 mL of 0.5 M KOH. The KOH was then titrated with 0.1 M HCl to an end point pH of 8.3 (Anderson, 1982; Alef, 1998). The cumulative amount of $\text{CO}_2\text{-C}$ evolved was calculated based on the measured daily $\text{CO}_2\text{-C}$ evolution rate.

Microbial Biomass

Microbial biomass was determined by fumigation-extraction (Vance et al., 1987) using a 7-d chloroform fumigation period as recommended by Sparling and Zhu (1993) for similar Western Australian soils. Briefly, 20 g (dry weight equivalent) of soil, either fumigated or nonfumigated, was mixed in a 1:4 ratio with 0.5 M K_2SO_4 , shaken for 1 h, and filtered through Whatman (Maidstone, UK) number 42 filter papers that had been prewashed with deionized water. Extracts were then stored at -18°C until analysis. Microbial biomass carbon (MB-C) was determined using high temperature oxidation (TOC-5000A; Shimadzu, Kyoto, Japan) and calculated as the total oxidizable carbon flush multiplied by a conversion factor ($k_{\text{EC}} = 2.64$; Vance et al., 1987). Microbial biomass nitrogen (MB-N) was analyzed by ninhydrin-positive compounds (NPC) (Brookes et al., 1985) and calculated as the NPC flush multiplied by a conversion factor ($k_{\text{EN}} = 3.5$; Sparling and Zhu, 1993).

Gross Nitrogen Transformations

Gross N mineralization rates were determined by ^{15}N isotopic pool dilution. The principle and assumptions of this technique have been reviewed elsewhere (Murphy et al., 2003). Briefly, this process involved first enriching the soil NH_4^+ pool with ^{15}N to increase the enrichment above natural abundance. The dilution of the ^{15}N enrichment in the NH_4^+ pool and the change in size of the NH_4^+ pool is then traced through time as organic matter is mineralized, releasing NH_4^+ at natural abundance. Gross N mineralization rates were determined on Days 3, 9, 16, 56, 81, and 142. In this study all soil incubations were conducted at 65 to 75% water holding capacity. For this reason we chose to deliver the ^{15}N label to soil using the simpler method of applying ^{15}N solution instead of $^{15}\text{NH}_3$ gas injection which does not alter the soil water content (Murphy et al., 1997). Previous studies (Murphy et al., 1999; Luxhøi et al., 2005) that have compared the $^{15}\text{NH}_3$ gas injector to ^{15}N solution addition found that there was no significant difference between measured gross N mineralization rates when the soil was moist. Soil (200 g) from each container was spread in a layer approximately 5 mm thick and 5 mL of $(^{15}\text{NH}_4)_2\text{SO}_4$ solution at 60 atom % ^{15}N was applied drop-wise over the surface. On Day 3 the ^{15}N solution was applied at a rate of 5 mg $\text{NH}_4^+\text{-N kg}^{-1}$ soil. All other sampling times received ^{15}N solution at a rate of 3 mg $\text{NH}_4^+\text{-N kg}^{-1}$. Immediately after labeling the soil was mixed thoroughly and put into 250-mL containers that were covered to minimize water loss and incubated at 15°C. At three time intervals after ^{15}N labeling (2, 24, and 48 h), 50 g of ^{15}N -labeled soil (dry wt. equivalent) was removed and the $\text{NH}_4^+\text{-N}$ and $\text{NO}_3^-\text{-N}$ extracted using 200 mL 0.5 M

K₂SO₄. Soil plus solution was shaken for 1 h, centrifuged at 2000 rpm for 5 min, and then filtered through Whatman number 41 filter papers using Buchner funnels under vacuum. Filtrate was collected and stored at -18°C until analysis. Soil remaining on the filter paper was washed twice with 200 mL of deionized water, dried (40°C), and ground using a ring-mill grinder.

All filtrates were analyzed for NH₄⁺-N and NO₃⁻-N + NO₂⁻-N on a Skalar (Breda, the Netherlands) San plus system continuous flow colorimetric analyzer. The diffusion method of Brooks et al. (1989) with slight modifications (incubated at 20°C for 7 d) was used to obtain separate ¹⁵N enrichments for both the NH₄⁺-N and NO₃⁻-N + NO₂⁻-N pools. A continuous flow system using a Tracermass Ion Ratio Mass Spectrometer and Roboprep preparation system (Europa PDZ, Crewe, UK) was used to determine ¹⁵N enrichments of the acidified diffusion discs and of 50-mg samples of the residual soil. Gross N mineralization rates were determined using the analytical equation of Kirkham and Bartholomew (1954). Net N mineralization rates were calculated as the difference between inorganic N values in extracts on subsequent sampling times divided by the time interval between the sampling periods.

The cumulative masses of CO₂-C evolved and cumulative gross N mineralized over the 142-d incubation period were estimated for each amendment based on linear extrapolation between the periodically measured time points. This approach has been used by other authors in similar studies (Andersen and Jensen, 2001; Ambus et al., 2002; Cookson et al., 2002) where it is not practical to measure C and/or N flux rates on a daily basis. The cumulative amount of N mineralized was predicted based on the CO₂-C evolution rates, using Eq. [3] from Murphy et al. (2003). A microbial efficiency of 0.4 (i.e., bacterial dominated population; Van Veen et al., 1985) was assumed and the C to N ratio of the organic amendment was used as the C to N ratio of the mineralizing pool [Eq. 1]. These predicted values were then compared with measured values.

$$\text{gross N mineralized} = [\text{CO}_2\text{-C evolved} / (1 - \text{microbial efficiency})] / \text{C to N ratio of amendment} \quad [1]$$

To increase the robustness of this prediction (i.e., from five to six amendments) we also included published CO₂-C evolution and gross N mineralization data for a grape marc amendment (Flavel et al., 2005). Chemical characteristics of the grape marc were as follows: total C = 42.6%, total N = 2.7%, NH₄⁺-N = 0.73 mg N kg⁻¹, NO₃⁻-N = 0.79 mg N kg⁻¹, ¹³C NMR carbon characterization (carbonyl C = 5.3%, O-aryl-C = 8.1%, aryl-C = 14.8%, di-O-alkyl-C = 11.3%, O-alkyl-C = 43.5%, alkyl-C = 17.0%), and ANKOM analysis (acid detergent fiber [ADF] = 48.7%, cellulose = 17.1%, lignin = 28.1%, ash = 3.5%).

Statistical Analysis

Statistical analysis was performed using SPSS Version 10 (Coakes and Steed, 2001). Repeated measure ANOVAs ($p < 0.05$) were used for time series data as detailed by Webster and Payne (2002). Spearman's correlation coefficients were used to determine significant ($p < 0.05$) relationships between amendment quality parameters and cumulative amounts of mineralized C or N from the amendments.

RESULTS

Soil Carbon Dioxide-Carbon Evolution

Soil CO₂-C evolution rates varied significantly among treatments (Fig. 1). The control (Fig. 1a) remained low (<2 mg CO₂-C kg⁻¹ d⁻¹) for the duration of the incuba-

tion. Treatment PP was initially much higher than the other amendments at >85 mg CO₂-C kg⁻¹ d⁻¹, then showed a rapid decline in CO₂-C evolution rates over time so that by Day 82 rates were of similar magnitude to the other amendments (Fig. 1b). GWCb had a higher CO₂-C evolution rate than GWCa; however, this difference was only significant ($p < 0.05$) on Days 3 and 37. The SBC treatment was also lower than GWCb. The CO₂-C evolution rates from GWCa, GWCb, and SBC treatment peaked on Day 51 and were 12.4, 15.6, and 10.4 mg CO₂-C kg⁻¹ d⁻¹, respectively. By Day 142 only PP and GWCa had CO₂-C evolution rates significantly different from the control soil.

By Day 70 PP treatment had respired CO₂-C equivalent to 36% of the C initially added in the amendment, and between Days 70 and 142 the further CO₂-C evolved equated to a further 10% of the added C. The other amendments (SBC, GWCa, GWCb, and VER) had evolved similar percentages of their total C by Day 70 (7.7, 7.3, 11.0, and 6.9%, respectively). The amount of CO₂-C evolved over the 142 d as a percentage of the C added with each amendment was in the order PP > GWCb ≥ GWCa ≥ SBC > VER.

Microbial Biomass

The MB-C in the control soil did not significantly change through the incubation and fluctuated between 41 and 85 mg C kg⁻¹ (Fig. 2a). However, MB-N significantly increased in the control between Day 0 (6 mg N kg⁻¹) and Day 9 (49 mg N kg⁻¹) and then declined rapidly to reach the original values by Day 108 (5 mg N kg⁻¹) (Fig. 2a). The MB-C in all amended soils was higher than in the control soil on all sampling times except in GWCa on Days 37 and 82. The MB-C in GWCb peaked at Day 23 (Fig. 2d), all other amended soils peaked at Day 16 (Fig. 2b, 2c, 2e, 2f) then decreased over the next two samplings. Microbial biomass C then remained relatively constant and at values similar to those at the start for each product except for GWCa, GWCb, and SBC, which increased again at the final samplings (Fig. 2c-2e). This late increase was not apparent in MB-N values (Fig. 2c-2e).

The amount of MB-N as a percentage of the total N available to the microbes (soil N + amendment N) was significantly higher in the PP-amended soils at 5.2%. There was no significant difference ($p > 0.05$) in MB-N as a percentage of the total N between the control (1.3%), GWCa (1.2%), and GWCb (1.8%) treatments or between SBC (2.2%), VER (2.4%), and GWCb. When averaged over the entire incubation the mean MB-C to MB-N ratio for each amendment was not significantly different and ranged from 5.8 to 6.9 for amendments, with 6.2 for the control soil.

Soil Inorganic Nitrogen Concentrations

With the exception of the PP treatment the soil NH₄⁺-N concentrations remained low (<5 mg NH₄⁺-N kg⁻¹) in amended soils for the duration of the incubation and were not significantly different from the control (Fig. 3). The PP treatment had a higher concentration of NH₄⁺-N between Days 3 and 51 which then decreased

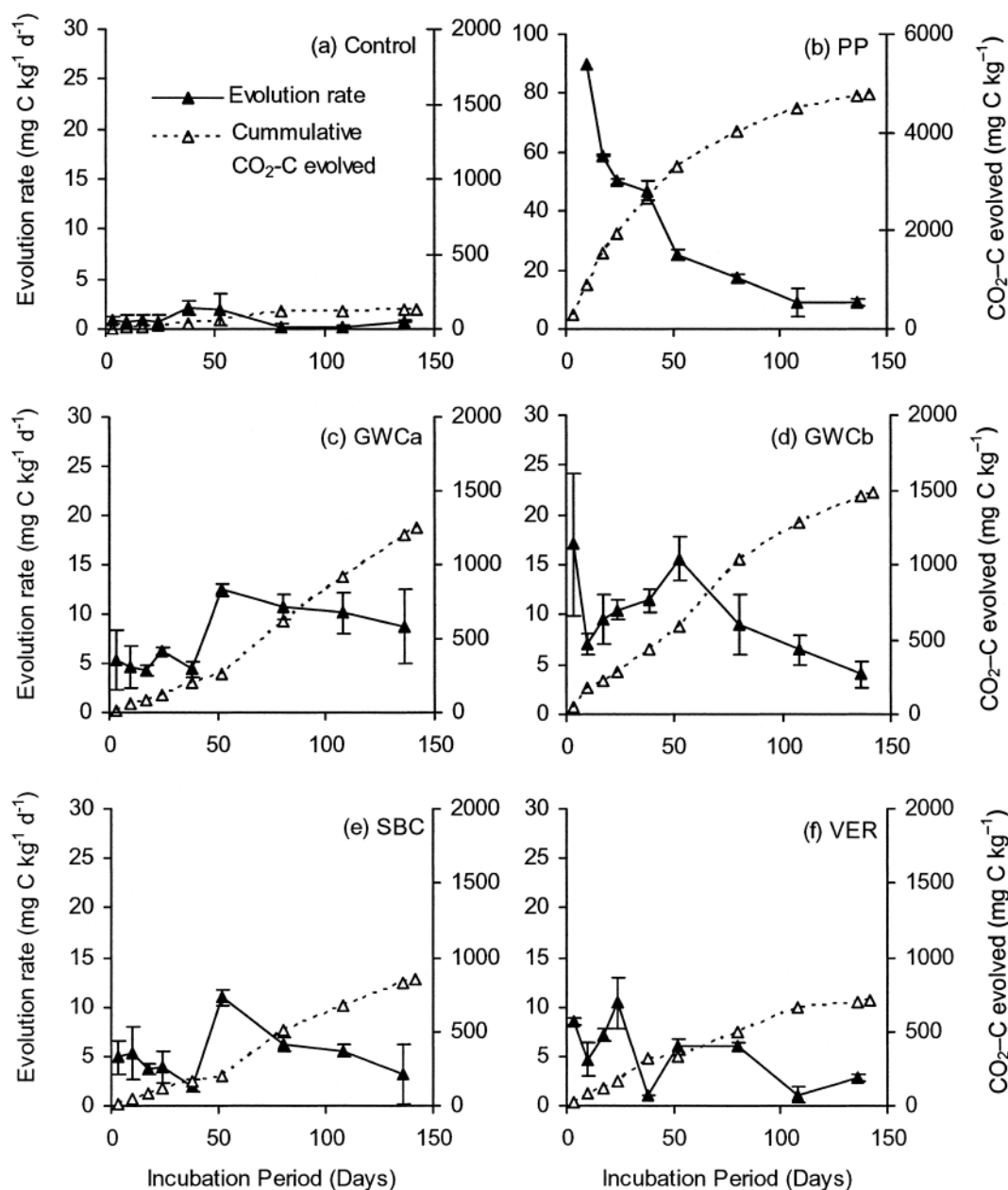


Fig. 1. Carbon dioxide (CO₂-C) evolution rates and cumulative amounts of CO₂-C evolved from amended soils (PP, pelletized poultry manure; GWCa, 18-wk green waste-based compost; GWCb, 24-wk green waste-based compost; SBC, straw-based compost; VER, vermi-cast). Bars indicate SEM; *n* = 4. Note the different scale on the PP graph.

as NO₃⁻-N increased (Fig. 3b). Nitrate N concentrations in the control and VER- and SBC-amended soils increased over the duration of the incubation (Fig. 3a, 3e, 3f). In GWCb-amended soils the NO₃⁻-N concentration was not significantly different from zero during the incubation period (Fig. 3d). GWCa was the only product in which NO₃⁻-N concentrations decreased with time (Fig. 3c).

Nitrogen Mineralization

Net N mineralization rates for the control soil were low throughout the incubation period and ranged from -0.01 to 0.3 mg N kg⁻¹ d⁻¹ with a mean of 0.11 (SEM = 0.04) mg kg d⁻¹. When averaged over the length of the

incubation period, the net N mineralization rates in all amendments were not significantly different from the control soil (data not shown).

Gross N mineralization rates in the amended soils were generally higher than the control soil; however, these differences were not always significant (Table 2). By Day 142 the cumulative amounts of gross N mineralized for each amendment were 109, 127, 133, 177, and 1052 mg N kg⁻¹ for GWGb, GWCa, VER, SBC, and PP, respectively. In the PP treatment the cumulative gross N mineralized equated to 137% of the N added in PP while in the other amendments the range was between 29 and 65% of the N originally incorporated within the amendment. There was a strong relationship

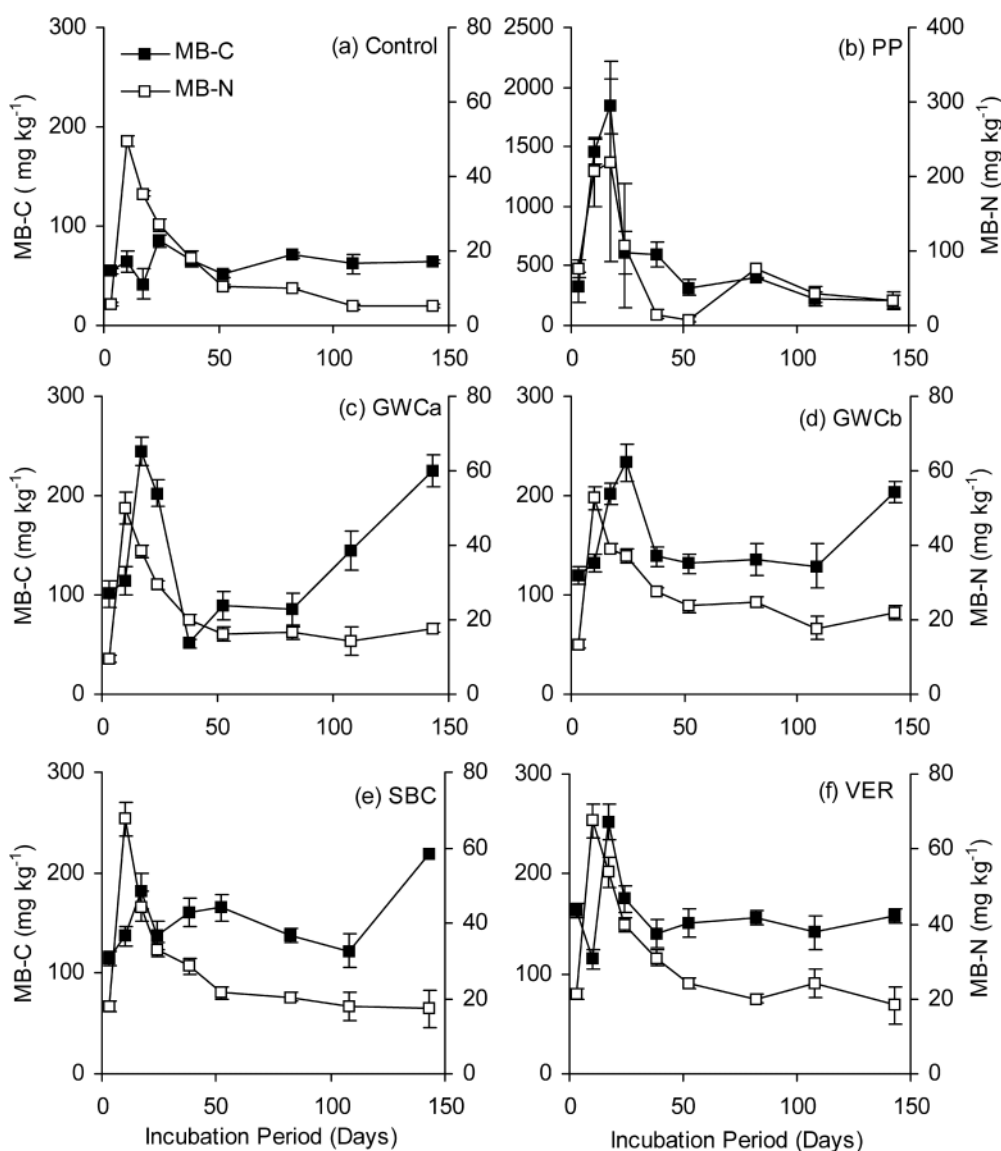


Fig. 2. Microbial biomass carbon (MB-C) and microbial biomass nitrogen (MB-N) from amended soils (PP, pelletized poultry manure; GWCa, 18-wk green waste-based compost; GWCb, 24-wk green waste-based compost; SBC, straw-based compost; VER, vermi-cast). Bars indicate SEM. Note the different scale on the PP graph.

($R^2 = 0.946$) between cumulative gross N mineralized and cumulative $\text{CO}_2\text{-C}$ evolved (Fig. 4a). There was also a strong relationship between measured and predicted gross N mineralized ($R^2 = 0.944$) (Fig. 4b) when Eq. [3] from Murphy et al. (2003) was used. However, predicted gross N mineralization rates were underestimated using this relationship (slope = 0.382).

Amendment Characteristics

The chemical characteristics of the amendments are given in Table 1. The $\text{NH}_4^+\text{-N}$ to $\text{NO}_3^-\text{-N}$ ratio was >0.16 in all amendments, with the exceptions of VER which was not significantly different from 0.16 and SBC which was <0.16 . The lignin to total N ratio of the products ranged between 5.7 for GWCb and 1.25 for PP. The spectra for the organic amendments differed mainly in the carbonyl-C (190–165 ppm), O-alkyl-C (140–110 ppm), and alkyl-C (45–0 ppm) regions (Table 3). As decomposition

proceeds the alkyl signal increases and the O-alkyl signal decreases (Webster et al., 2001). Ranking amendments in order of decomposition based on either alkyl-C or O-alkyl-C contents gave the same result, $\text{PP} < \text{GWGb} < \text{GWCa} < \text{SBC} < \text{VER}$, with PP being the least decomposed. The green waste-based composts (GWCa and GWGb) had the same alkyl-C to O-alkyl-C ratio of 0.19, PP had a ratio of 0.22, SBC a ratio of 0.41, and VER had the highest value with 0.44. Ranking amendments on their ash content also indicated that PP was the least stable product ($\text{PP} < \text{GWGb} < \text{GWCa} < \text{VER} < \text{SBC}$).

Influence of Amendment Quality on Carbon and Nitrogen Mineralization

Spearman's rho correlations indicated that the cumulative amounts of gross N mineralized were significantly correlated to the products initial total N ($r = 0.919$, $p = 0.003$) and total C content ($r = 0.929$, $p = 0.003$) and

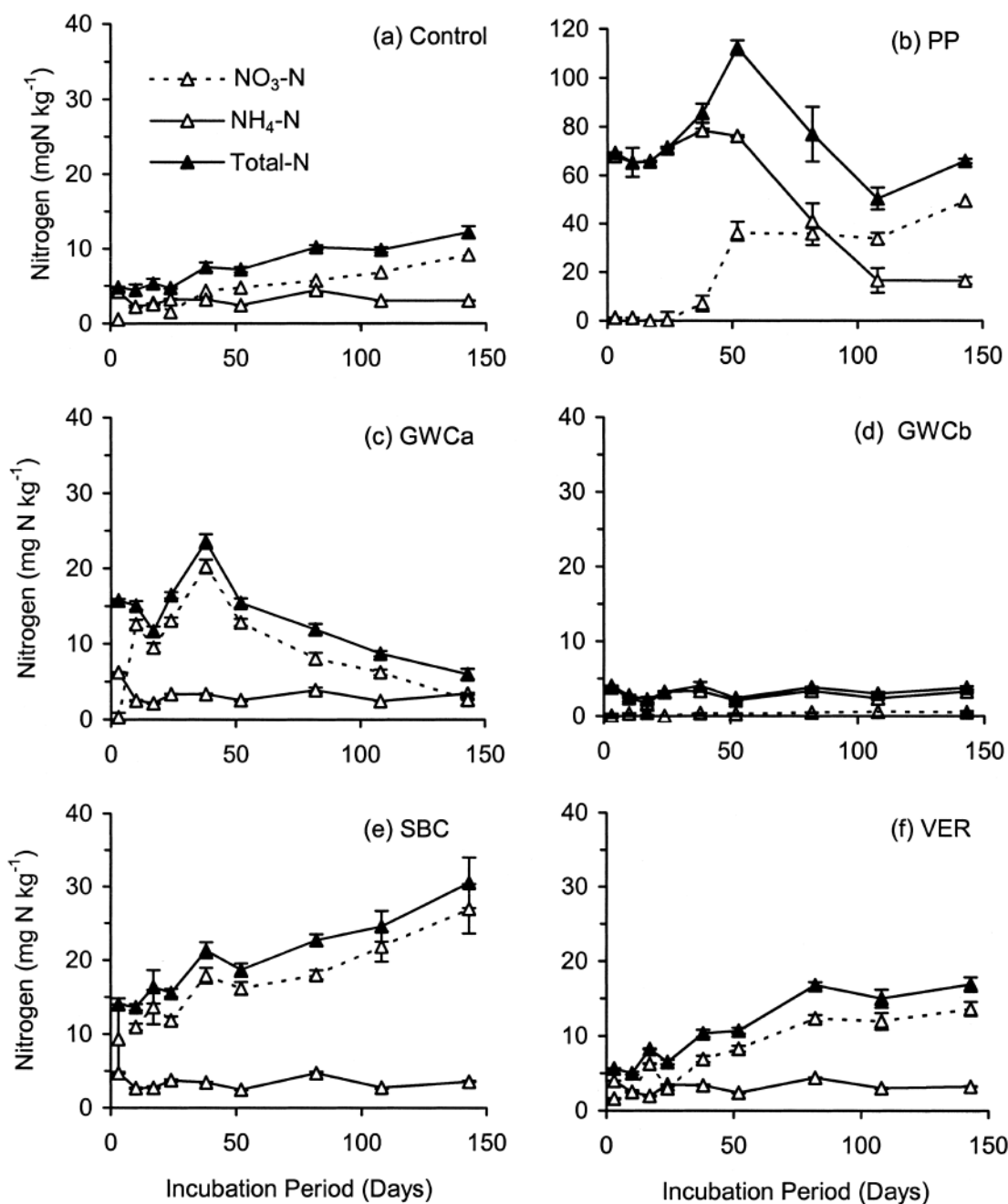


Fig. 3. Inorganic N traces for incubated soils (PP, pelletized poultry manure; GWCa, 18-wk green waste-based compost; GWCb, 24-wk green waste-based compost; SBC, straw-based compost; VER, vermi-cast). Bars indicate SEM; $n = 4$. Note the different scale on the PP graph.

Table 2. Analytically calculated gross N mineralization rate ($n = 4$).

Treatment†	Gross N mineralization					
	Day					
	3	9	16	37	82	142
	mg N kg ⁻¹ d ⁻¹					
Control	0.77 (0.08) ab‡	0.15 (0.08) a	0.06 (0.03) a	0.27 (0.04) a	2.52 (0.39) ab	0.30 (0.18) a
PP	27.17 (6.69) c	13.59 (4.62) e	2.91 (1.74) abc	7.98 (2.51) b	12.95 (3.02) c	9.94 (0.97) c
GWCa	0.99 (0.26) ab	1.15 (0.22) bc	0.79 (0.16) bc	0.62 (0.56) a	2.93 (0.35) ab	1.77 (0.24) b
GWGb	0.51 (0.11) a	0.66 (0.31) ab	1.12 (0.08) b	5.76 (2.12) b	2.01 (0.21) a	1.63 (0.08) b
SBC	1.10 (0.17) b	2.13 (0.52) cd	0.84 (0.17) bc	0.51 (0.08) a	2.62 (0.10) ab	0.87 (0.49) ab
VER	0.72 (0.16) ab	2.62 (0.04) d	0.57 (0.19) c	0.56 (0.15) a	3.40 (0.49) b	1.42 (0.23) b

‡ Numbers in parentheses are standard error of the mean. Values with the same letter are not significantly different based on 95% confidence intervals.

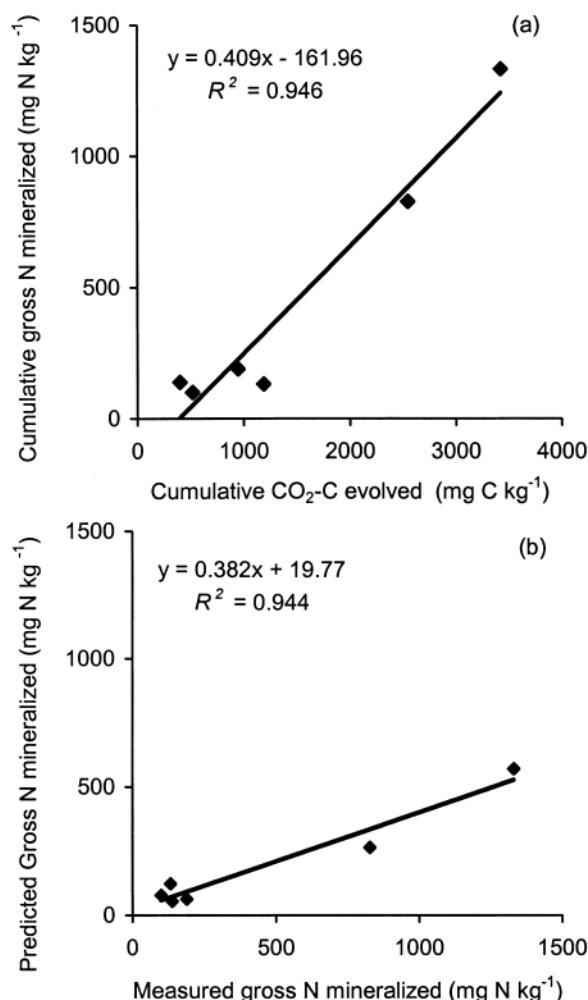


Fig. 4. (a) Relationship between cumulative CO₂-C evolved and cumulative gross N mineralized. (b) Measured versus predicted amounts of cumulative gross N mineralized.

also to the cellulose ($r = 0.929$, $p = 0.003$) and lignin ($r = 0.893$, $p = 0.007$) content. There was a negative relationship to the total ash ($r = -0.964$, $p = 0.000$) and NO₃⁻-N ($r = -0.928$, $p = 0.008$) contents of the amendments. There were significant relationships between the cumulative amounts of C mineralized from the amendments and the initial total C content ($r = 0.943$, $p = 0.005$), cellulose ($r = 0.851$, $p = 0.014$), NH₄⁺-N ($r = 0.886$, $p = 0.019$), and ash content ($r = -0.900$, $p = 0.037$) of the amendments. No other param-

eters tested (including the ratios of lignin to total N, NH₄⁺-N to NO₃⁻-N, and alkyl-C to O-alkyl-C) indicated significant relationships ($p < 0.05$) with either the cumulative amounts of C or N mineralized.

DISCUSSION

Processes such as composting cause organic compounds to stabilize reducing the proportion of C and N held in soluble form (Gigliotti et al., 2002). Mature composts that have undergone the complete composting process have a much higher proportion of stable materials than immature or noncomposted products (Bernal et al., 1998c). Hence, the application of composted amendments (as opposed to raw manures and noncomposted amendments) to soil can reduce the immediate leaching risks when crops are small and N demand is low. A C to N ratio of <12 has been suggested to indicate a mature compost product (Bernal et al., 1998b); by this standard PP, SBC, and VER should all behave in a similar manner, as "mature" amendments. While VER behaved as a mature amendment, the others did not, indicating that the total C to N ratio did not always predict the behavior of the organic amendments in soil. This is supported by other studies that suggest it is the quality of C and N rather than the total amounts that have the greatest influence on decomposition (Kogel-Knaber, 2002; Vigil and Kissel, 1995; Wang et al., 2004b).

Other measures suggested to indicate mature and stable composts include cumulative CO₂-C evolution values equivalent to <25% of the total C applied and a NH₄⁺-N to NO₃⁻-N ratio of <0.16 (Bernal et al., 1998b, 1998c). Applying these parameters to the tested amendments indicated that the PP treatment was the most immature and unstable product. Straw-based compost was the only amendment with a NH₄⁺-N to NO₃⁻-N ratio of <0.16; however, VER was not significantly different from 0.16. The CO₂-C evolution values also indicate SBC and VER were the most stable amendments tested. The ash content generally increases and the CO₂-C evolution rate decreases as the composting process continues and a more stable product is achieved (Wang et al., 2004a). Again these parameters indicated that the PP treatment was the least stable product, followed by GWCa and GWCb and then SBC and VER. Carbon characterization and ¹³C NMR spectra also suggest that PP was the least stable treatment as it had a higher proportion of degradable material than the other products. The "matured" compost, GWCb, was in fact

Table 3. Characterization of the total carbon as determined by the distribution of the ¹³C nuclear magnetic resonance (NMR) signal for each amendment.

Treatment†	Carbonyl-C (190–165 ppm)	O-aryl-C (165–140 ppm)	Aryl-C (140–110 ppm)	di-O-alkyl-C (110–90 ppm)	O-alkyl-C (90–45 ppm)	Alkyl-C (45–0 ppm)
% of the spectra under each chemical shift region						
PP	5.2	7.5	8.6	13.5	53.3	11.8
GWCa	4.5	7.7	12.7	14.1	51.4	9.6
GWGb	3.4	7.3	12.5	13.5	53.2	10.2
SBC	8.2	7.9	12.8	10.7	42.9	17.5
VER	6.4	8.1	13.9	11.1	42.1	18.4

† PP, pelletized poultry manure; GWCa, 18-wk green waste-based compost; GWGb, 24-wk green waste-based compost; SBC, straw-based compost; VER, vermi-cast.

more like an immature, unstable product than the younger compost, GWCa. While every attempt was made to ensure that the two green waste-based composts were the same (with the exception of a maturation phase), the results of the chemical analysis suggest that this was not achieved. GWCb had a higher C to N ratio, higher content of O-alkyl-C, lower content of alkyl-C, lower ash content, higher cellulose content, higher $\text{NH}_4^+\text{-N}$ to $\text{NO}_3^-\text{-N}$ ratio, and higher lignin to total N ratio than GWCa. GWCb also had initially higher $\text{CO}_2\text{-C}$ evolution rates and higher MB-N concentrations than GWCa, again indicating a less stable product. We suggest that this is a reflection of the variability in the waste source used (possibly different combinations of plant types in the green waste collected), resulting in differences in the composition of the two composts that override any effect of the extended "maturation" phase.

Soils amended with less stabilized products had higher MB-C concentrations than those with more stabilized materials which has also been found by other authors (Paul and Solaiman, 2004; Sanchez-Monedero et al., 2004). The initial flush in microbial biomass is partly due to microbes added with the product (Perucci, 1992) with the early stages of the composting process having higher microbial biomass than later stages (Mondini et al., 1997; Sanchez-Monedero et al., 2004). However, the major cause for an increase in the microbial populations in amended soils is due to the increase in available C as a food source to allow populations to quickly multiply given many of the species introduced with the compost die soon after amendment (Sastre et al., 1996). Sanchez-Monedero et al. (2004) found that MB-N concentrations from matured compost samples reached concentrations similar to the control samples after 60 d of incubation and that MB-N measurements were erratic due to the concentration of inorganic N present in the soil. In general none of the amendments tested in this study had MB-N or MB-C concentrations decline to values of the control soil. Gigliotti et al. (2002) concluded that urban waste compost offered no labile organic matter that would serve as an energy source for microorganisms and suggest that less processed materials such as sewage sludge or pig slurry are a better microbial food supply. Our results indicate that the composted products did have less available organic matter than the less processed product (e.g., PP). However, all the amendments tested were able to supply enough organic matter for microbial energy for at least 142 d as evident in the both the microbial biomass and $\text{CO}_2\text{-C}$ evolution traces (Fig. 1 and 2).

The amount of lignin and recalcitrant C has been suggested to reduce the rate of N mineralization (Vigil and Kissel, 1995), as this C is resistant to most forms of microbial attack (Kogel-Knaber, 2002). Ranking products in the order of initial lignin content did not correspond to the same ranking as gross N mineralization rates for Day 3, nor to the cumulative amount of gross N mineralized over the length of the incubation. Wang et al. (2004b) found the lignin to total N ratio and the ratios between carbonyl-C, aryl-C, and O-aryl-C contents to be significantly correlated to cumulative $\text{CO}_2\text{-C}$

evolved after 160 d of incubating various plant residues, yet there were no significant correlations for those parameters in this study. The lignin to total N ratios in this study were not significantly related to the amount of N mineralized as has also been reported by other authors (Rowell et al., 2001). Rowell et al. (2001) found the alkyl-C contents were correlated with net N mineralization. We found no relationship between the alkyl-C values and either net or gross N mineralization. Of the C quality parameters we tested, the cellulose, lignin, and ash contents were the only ones to have an influence on amounts of gross N mineralized. The total C and total N contents of the amendments both had significant effects on N mineralization. Booth et al. (2005) also suggest that the total C and total N content of soils will have a large impact on gross N fluxes.

Given that the mineralization processes involved in the N cycle are biotic it would stand to reason these processes would be higher in soils with a higher microbial biomass and activity (Bengtsson et al., 2003), providing other factors are the same. In the PP-amended soils the highest gross N mineralization rates were obtained on the same days as the high MB-C and $\text{CO}_2\text{-C}$ evolution rates were recorded (Table 2, Fig. 1 and 2); however, this was not the case in all treatments. This may be due to the quality of C and N in the organic amendment, or because the proportion of the microbial biomass that is active in each amended soil is different. Microbial biomass as measured by fumigation-extraction is a measure of the total biomass, not just the active fraction (Ross, 1991), which may be why in this and other studies (Bengtsson et al., 2003) it has been poorly correlated to N mineralization rates.

Cumulative gross N mineralized in the PP treatment was greater than 100% of the N contained within the amendment. This could reflect the fact that there is rapid recycling of the same NH_4^+ molecules through mineralization-immobilization turnover within the closed system. Previously, Murphy et al. (1998) calculated a residence time of <5 d for the NH_4^+ pool in a Western Australian loamy sand when moisture was not limiting. Given the preference for microbial immobilization of NH_4^+ , compared to NO_3^- , and the typically small NH_4^+ pool size in control soils of the current study, it is to be expected that the residence time of NH_4^+ is short. The ^{15}N isotopic pool dilution technique quantifies a flux rate over a period of only a few days (typically <3 d) to avoid remineralization but does not distinguish between N that is being mineralized directly from the organic residue source and N that is derived from the microbial biomass turning over and releasing the previously immobilized residue N. Thus during a long-term incubation study in a closed system it is likely that the same N becomes "counted" more than once as mineralization-immobilization turnover and subsequent remineralization occurs. Murphy et al. (2003) highlighted this issue with data from Cookson et al. (2002) where incorporation of plant residues containing 403 mg N kg^{-1} resulted in cumulative gross N mineralization rates of nearly 800 mg N kg^{-1} . Recoveries above 100% have also been reported in compost sludge incubation studies (Ambus et al., 2002). Another issue

with determining accurate gross N mineralization rates in the PP treatment is the high unlabeled $\text{NH}_4^+\text{-N}$ pool size ($>60 \text{ mg NH}_4^+\text{-N}$) before ^{15}N solution addition during the early stage of the incubation period (3–37 d). Luxhøi et al. (2005) illustrated that gross N mineralization rates are likely to be overestimated in situations where small additions of $^{15}\text{NH}_4^+$ are made to soils with high (approximately $>30 \text{ mg NH}_4^+\text{-N kg}^{-1}$) ambient $^{14}\text{NH}_4^+$ pool size. This could also contribute to a cumulative gross N mineralization rate above 100%. However, PP fits the linear relationship between $\text{CO}_2\text{-C}$ evolved and gross N mineralization (Fig. 4a) suggesting that gross N mineralization rates were not grossly overestimated.

Although the overall net N mineralization rates were similar for all treatments, the amount and timing of inorganic N release were different. In terms of application of amendments to horticultural soils an amendment such as GWCB, which had lower total inorganic N values than the control, would have to be carefully managed to ensure adequate inorganic N was in the soil at the time of plant N demand. Managers considering application of a relatively unprocessed amendment like PP need to know that it is likely to increase microbial biomass size and activity, and release large amounts of N into the soil system soon after application but not contribute to the long-term buildup of SOM. This is particularly likely in irrigated systems of warm environments, such as those in Western Australia, which provide ideal microbial growth conditions. As organic amendments are applied to these systems before planting the N demand by the crop is low and the early release of $\text{NH}_4^+\text{-N}$ is unlikely to be used, hence potentially nitrified and at risk of being lost from the systems via leaching. Thus the rate and timing of PP applications need to consider the N requirements of the given crop to ensure efficient use of N and minimize potential $\text{NO}_3^-\text{-N}$ leaching hazards into surrounding urban water sources, especially where algal blooms are known to occur.

CONCLUSIONS

It is generally accepted that C and N cycling within soil is intrinsically linked and that the degree to which a soil amendment will impact on these cycles will depend on the quality of that amendment. Here we found a significant relationship between C mineralization and gross N mineralization. While net N mineralization rates were not significantly different between treatments, there were differences in gross N mineralization rates reflecting differences in the magnitude of N cycling between amendments. Previous studies have suggested the use of ^{13}C NMR shift regions or their associated ratios as a predictor of the N release capacity of a residue. We found no such relationships to $\text{CO}_2\text{-C}$ evolution and gross N mineralization rates. The amount of gross N mineralized was, however, significantly related to the total C and N, cellulose, lignin, ash, and $\text{NO}_3^-\text{-N}$ contents of the amendments. There were also significant relationships ($p < 0.05$) between the cumulative amounts of $\text{CO}_2\text{-C}$ evolved and the total C and ash content of

the amendments. Further research is required to provide quantifiable indicators of maturity status, as it is imperative that landholders can manage the application of organic amendments to soil through accurate knowledge of their maturity status and thus nutrient release characteristics.

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REFERENCES

- Alef, K. 1998. Soil respiration. p. 213–219. In K. Alef and P. Nannipieri (ed.) *Methods in applied soil microbiology and biochemistry*. Academic Press, Sydney.
- Ambus, P., L.K. Kure, and E.S. Jensen. 2002. Gross N transformation after application of household compost and domestic sewage sludge on agricultural soils. *Agronomie* 22:723–730.
- Andersen, M.K., and L.S. Jensen. 2001. Low soil temperature effects on short-term gross N mineralization-immobilization turnover after incorporation of a green manure. *Soil Biol. Biochem.* 33:511–521.
- Anderson, J.P.E. 1982. Soil respiration. p. 831–871. In A.L. Page et al. (ed.) *Methods of soil analysis*. Part 2. 2nd ed. Agron. Monogr. 9. ASA and SSSA, Madison, WI.
- Bengtsson, G., P. Bengtson, and K.F. Mansson. 2003. Gross nitrogen mineralization-, immobilization-, and nitrification rates as a function of soil C/N ratio and microbial activity. *Soil Biol. Biochem.* 35:143–154.
- Bernal, M.P., A.F. Navarro, M.A. Sanchez-Monedero, A. Roig, and J. Cegarra. 1998a. Influence of sewage sludge compost stability and maturity on carbon and nitrogen mineralization in soil. *Soil Biol. Biochem.* 30:305–313.
- Bernal, M.P., C. Paredes, M.A. Sanchez-Monedero, and J. Cegarra. 1998b. Maturity and stability parameters of composts prepared with a wide range of organic wastes. *Bioresour. Technol.* 63:91–99.
- Bernal, M.P., C. Sanchez-Monedero, C. Paredes, and A. Roig. 1998c. Carbon mineralization from organic wastes at different composting stages during their incubation with soil. *Agric. Ecosyst. Environ.* 69:175–189.
- Booth, M.S., J.M. Stark, and E. Rastetter. 2005. Controls on gross nitrogen cycling rates in terrestrial ecosystems: A synthesis and analysis of data from the literature. *Ecol. Monogr.* 75:139–158.
- Brookes, P.C., A. Landman, G. Pruden, and D.S. Jenkinson. 1985. Chloroform fumigation and the release of soil nitrogen: A rapid direct extraction method to measure microbial biomass nitrogen in soil. *Soil Biol. Biochem.* 17:837–842.
- Brooks, P.D., J.M. Stark, B.B. McInteer, and T. Preston. 1989. Diffusion method to prepare soil extracts for automated nitrogen-15 analysis. *Soil Sci. Soc. Am. J.* 53:1707–1711.
- Coakes, S.J., and L.G. Steed. 2001. SPSS: Analysis without anguish: Version 10.0 for Windows. John Wiley Publ., Brisbane.
- Cook, D., I. Dadour, N. Keals, and R. Paulin. 1997. Stable flies on the Swan Coastal Plain. *J. Agric. West. Aust.* 38:58–61.
- Cookson, W.R., I.S. Cornforth, and J.S. Rowarth. 2002. Winter soil temperature (2–15°C) effects on nitrogen transformations in clover green manure amended or unamended soils: A laboratory and field study. *Soil Biol. Biochem.* 34:1401–1415.
- Flavel, T.C., D.V. Murphy, B.M. Lalor, and I.R.P. Fillery. 2005. Gross N mineralization rates after the application of composted grape marc to soil. *Soil Biol. Biochem.* 37:1397–1400.
- Gabrielle, B., J. De-Silveira, S. Houot, and C. Francou. 2004. Simulating urban waste compost effects on carbon and nitrogen dynamics using a biochemical index. *J. Environ. Qual.* 33:2333–2342.
- Gigliotti, G., K. Kaiser, G. Guggenberger, and L. Haumaier. 2002. Differences in the chemical composition of dissolved organic matter

- from waste material of different sources. *Biol. Fertil. Soils* 36: 321–329.
- Gilmour, J.T. 1998. Carbon and nitrogen mineralization during co-utilization of biosolids and composts. p. 89–112. *In* S. Brown, J.S. Angle, and L. Jacobs (ed.) *Beneficial co-utilization of agricultural, municipal and industrial by-products*. Kluwer Academic Publ., Dordrecht, the Netherlands.
- Hadas, A., L. Kautsky, M. Goek, and E.E. Kara. 2004. Rates of decomposition of plant residues and available nitrogen in soil, related to residue composition through simulation of carbon and nitrogen turnover. *Soil Biol. Biochem.* 36:255–266.
- Hutchison, J. (ed.) 2002. *Code of Practice: For environmentally sustainable vegetable and potato production in Western Australia*. Quality Press, Perth.
- Kirkham, D., and W.V. Bartholomew. 1954. Equations for following nutrient transformations in soil utilizing tracer data. *Soil Sci. Soc. Am. Proc.* 18:33–34.
- Kogel-Knabner, I. 2002. The macromolecular organic composition of plant and microbial residues as inputs to soil organic matter. *Soil Biol. Biochem.* 34:139–162.
- Luxhøi, J., S. Recous, I.R.P. Fillery, D.V. Murphy, and L.S. Jensen. 2005. Comparison of $^{15}\text{NH}_4^+$ pool dilution techniques to measure gross N fluxes in a coarse textured soil. *Soil Biol. Biochem.* 37: 569–572.
- Mamo, M., C.J. Rosen, and T.R. Halbach. 1999. Nitrogen availability and leaching from soil amended with municipal solid waste compost. *J. Environ. Qual.* 28:1074–1082.
- Mondini, C., A. Sanchez-Monedero, L. Letita, G. Bragato, and M. De Nobli. 1997. Carbon and ninhydrin-reactive nitrogen of the microbial biomass in rewetted compost samples. *Commun. Soil Sci. Plant Anal.* 23:113–122.
- Murphy, D.V., A. Bhogal, M.A. Shepherd, K.W.T. Goulding, S.C. Jarvis, D. Barraclough, and J.L. Gaunt. 1999. Comparison of ^{15}N labeling methods to measure gross N mineralization. *Soil Biol. Biochem.* 31:2015–2024.
- Murphy, D.V., I.R.P. Fillery, and G.P. Sparling. 1997. Method to label soil cores with $^{15}\text{NH}_3$ gas as a prerequisite for ^{15}N isotopic dilution and measurement of gross N mineralization. *Soil Biol. Biochem.* 29:1731–1741.
- Murphy, D.V., I.R.P. Fillery, and G.P. Sparling. 1998. Seasonal fluctuations in gross N mineralization, ammonium consumption, and microbial biomass in a Western Australian soil under different land uses. *Aust. J. Agric. Res.* 49:523–535.
- Murphy, D.V., S. Recous, E.A. Stockdale, I.R.P. Fillery, L.S. Jensen, D.J. Hatch, and K.W.T. Goulding. 2003. Gross nitrogen fluxes in soil: Theory, measurement and application of ^{15}N pool dilution techniques. *Adv. Agron.* 79:69–118.
- Nicolardot, B., S. Recous, and B. Mary. 2001. Simulation of C and N mineralization during crop residue decomposition: A simple dynamic model based on the C: N ratio of the residues. *Plant Soil* 228:83–103.
- Paul, G.C., and A.R.M. Solaiman. 2004. Changes in microbial biomass carbon and nitrogen in upland sugarcane soil amended with different organic materials. *Commun. Soil Sci. Plant Anal.* 35:2433–2447.
- Perucci, P. 1992. Enzyme activity and microbial biomass in a field soil amended with municipal refuse. *Biol. Fertil. Soils* 14:54–60.
- Ross, D.J. 1991. Microbial biomass in a stored soil: A comparison of different estimation procedures. *Soil Biol. Biochem.* 23:1005–1007.
- Rowell, D.M., C.E. Prescott, and C.M. Preston. 2001. Decomposition and nitrogen mineralization from biosolids and other organic materials: Relationship with initial chemistry. *J. Environ. Qual.* 30: 1401–1410.
- Sanchez-Monedero, M.A., C. Mondini, M. de Nobili, L. Leita, and A. Roig. 2004. Land application of biosolids. Soil response to different stabilization degree of the treated organic matter. *Waste Manage.* 24:325–332.
- Sastre, I., M.A. Vicente, and M.C. Lobo. 1996. Influence of the application of sewage sludges on soil microbial activity. *Bioresour. Technol.* 57:19–23.
- Sikora, L.J., and R.A.K. Szmids. 2001. Nitrogen sources, mineralization rates, and nitrogen nutrition benefits to plants from composts p. 287–305. *In* P.J. Stoffella and B.A. Kahn (ed.) *Compost utilization in horticultural cropping systems*. Lewis Publ., New York.
- Smernik, R.J., J.A. Baldock, J.M. Oades, and A.K. Whittaker. 2002. Determination of $T_{1\rho}\text{H}$ relaxation rates in charred and uncharred wood and consequences for NMR quantitation. *Solid State Nucl. Magn. Reson.* 22:50–70.
- Smernik, R.J., and J.M. Oades. 2000a. The use of spin counting for determining quantification in solid state ^{13}C NMR spectra of natural organic matter. 1. Model systems and the effects of paramagnetic impurities. *Geoderma* 96:101–129.
- Smernik, R.J., and J.M. Oades. 2000b. The use of spin counting for determining quantification in solid state ^{13}C NMR spectra of natural organic matter. 2. HF-treated soil fractions. *Geoderma* 96:101–129.
- Sparling, G., and C. Zhu. 1993. Evaluation and calibration of biochemical methods to measure microbial biomass C and N in soils from Western Australia. *Soil Biol. Biochem.* 25:1793–1801.
- Stoffella, P.J., and B.A. Kahn (ed.) 2001. *Compost utilization in horticultural cropping systems*. Lewis Publ., New York.
- Vance, E.D., P.C. Brookes, and D.S. Jenkinson. 1987. An extraction method for measuring soil microbial biomass C. *Soil Biol. Biochem.* 19:703–707.
- Van Veen, J.A., J.N. Ladd, and M. Amato. 1985. Turnover of carbon and nitrogen through the microbial biomass in a sandy loam and clay soil incubated with (^{14}C (U)) glucose and (^{15}N) (NH_4) $_2\text{SO}_4$ under different moisture regimes. *Soil Biol. Biochem.* 17:747–756.
- Vigil, M.F., and D.E. Kissel. 1995. Rate of nitrogen mineralized from incorporated crop residues as influenced by temperature. *Soil Sci. Soc. Am. J.* 59:1636–1644.
- Wang, P., C.M. Chang, W.E. Watson, W.A. Dick, Y. Chen, and H.A.J. Hoitink. 2004a. Maturity indices for composted dairy and pig manures. *Soil Biol. Biochem.* 36:767–776.
- Wang, W.J., J.A. Baldock, R.C. Dalal, and P.W. Moody. 2004b. Decomposition dynamics of plant materials in relation to nitrogen availability and biochemistry determined by NMR and wet-chemical analysis. *Soil Biol. Biochem.* 36:2045–2058.
- Webster, E.A., D.W. Hopkins, J.A. Chudek, S.F.I. Haslam, M. Simek, and T. Picek. 2001. The relationship between microbial carbon and the resource quality of soil carbon. *J. Environ. Qual.* 30:147–150.
- Webster, R., and R.W. Payne. 2002. Analysing repeated measurements in soil monitoring and experimentation. *Eur. J. Soil Sci.* 53:1–13.
- Wolkowski, R.P. 2003. Nitrogen management considerations for land-spreading municipal solid waste compost. *J. Environ. Qual.* 32: 1844–1850.