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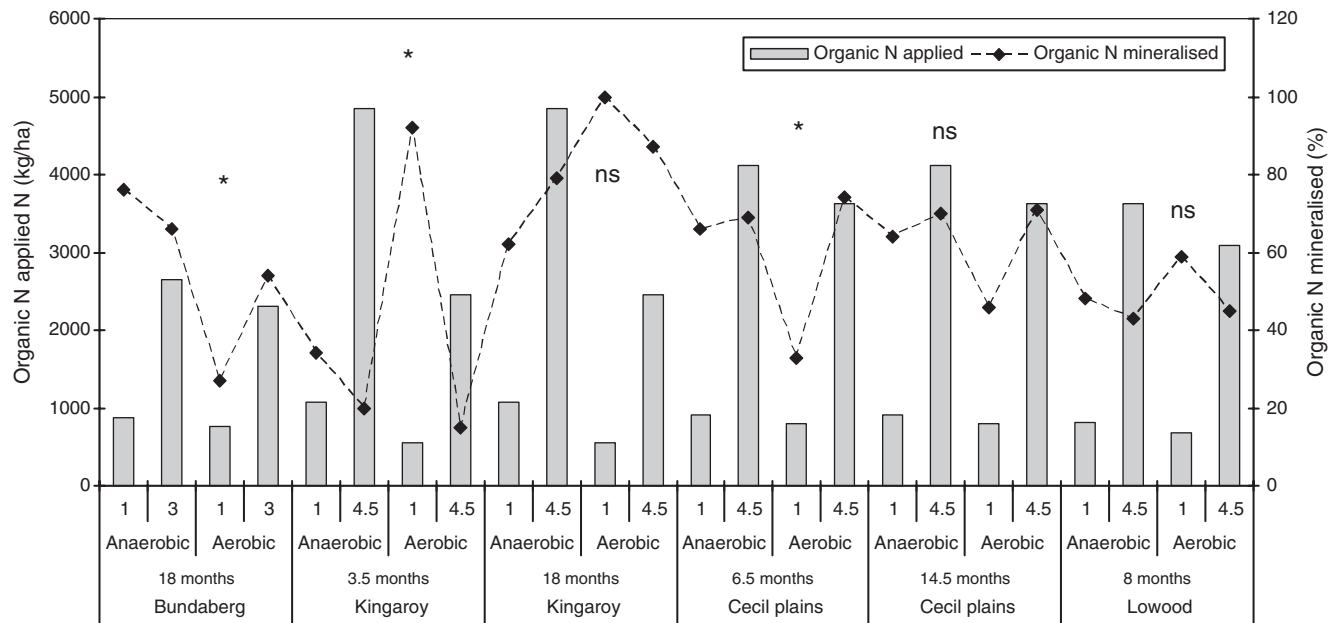
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The Figure 5 as published is incorrect. There is no label on the 2nd *y*-axis for % N mineralised in the figure. The authors regret this omission. The correct Figure 5 is below.

Estimating mineralisation of organic nitrogen from biosolids and other organic wastes applied to soils in subtropical Australia

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Abstract.   One major benefit of land application of biosolids is to supply nitrogen (N) for agricultural crops, and understanding mineralisation processes is the key for better N-management strategies. Field studies were conducted to investigate the process of mineralisation of three biosolids products (aerobic, anaerobic, and thermally dried biosolids) incorporated into four different soils at rates of 7–90 wet t/ha in subtropical Queensland. Two of these studies also examined mineralisation rates of commonly used organic amendments (composts, manures, and sugarcane mill muds).

Organic N in all biosolids products mineralised very rapidly under ambient conditions in subtropical Queensland, with rates much faster than from other common amendments. Biosolids mineralisation rates ranged from 30 to 80% of applied N during periods ranging from 3.5 to 18 months after biosolids application; these rates were much higher than those suggested in the biosolids land application guidelines established by the NSW EPA (15% for anaerobic and 25% for aerobic biosolids). There was no consistently significant difference in mineralisation rate between aerobic and anaerobic biosolids in our studies. When applied at similar rates of N addition, other organic amendments supplied much less N to the soil mineral N and plant N pools during the crop season.

A significant proportion of the applied biosolids total N (up to 60%) was unaccounted for at the end of the observation period. High rates of N addition in calculated Nitrogen Limited Biosolids Application Rates (850–1250 kg N/ha) resulted in excessive accumulation of mineral N in the soil profile, which increases the environmental risks due to leaching, runoff, or gaseous N losses. Moreover, the rapid mineralisation of the biosolids organic N in these subtropical environments suggests that biosolids should be applied at lower rates than in temperate areas, and that care must be taken with the timing to maximise plant uptake and minimise possible leaching, runoff, or denitrification losses of mineralised N.

Additional keywords: biosolids, leaching, mineralisation, nitrogen loss, runoff.

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Introduction

Land application of biosolids has received increased attention in the last two decades due to the ability of biosolids to replenish reduced soil organic matter, supply nutrients—mainly nitrogen (N) and phosphorus (P) and some essential micronutrients, such as zinc (Zn) and copper (Cu)—to plants, improve soil structure and water-holding capacity, and have beneficial effects on microbial biomass and activity (Eriksen *et al.* 1999; Leifeld *et al.* 2002; Bell *et al.* 2004).

The added mineral and organic N can affect N dynamics in soil–plant ecosystems. Most of the biosolids N is in the organic form (Pu *et al.* 2008), and in most cases, organic N can only be available to plants after it is mineralised to ammonium (NH_4) and nitrate (NO_3) by microorganisms in the soil. However,

anaerobically digested biosolids can contain significant proportions of total N in the form of NH_4-N (Pu *et al.* 2008), which can be vulnerable to loss through ammonia volatilisation during, and immediately after, application and incorporation.

Transformations of biosolids N, particularly mineralisation of the large organic N fraction, need to be estimated for better N-management strategies to increase crop production and minimise the potential for off-site movement by leaching and runoff. However, transformations of the applied N under field conditions largely depend on the composition of such applied materials, environmental conditions (such as rainfall and temperature), and the types and properties of the soils to which the biosolids are applied, as well as the agricultural production systems. The reported mineralisation rate under

field conditions ranges between 15 and 55% of the biosolids-sourced organic N during the first year after application (Binder *et al.* 2002; Robinson *et al.* 2002; Adegbidi and Briggs 2003; Vieira *et al.* 2005; Mendoza *et al.* 2006).

In most cases, the amount of biosolids applied in field situations is calculated from algorithms designed to elucidate a Nitrogen Limited Biosolids Application Rate (NLBAR), with one of the key factors in this algorithm being the rate at which the organic fraction of biosolids N is mineralised. In some states of Australia, commercial biosolids application rates are calculated using the methods recommended in the Biosolids Guidelines provided by the Environmental Protection Agency of New South Wales (NSW EPA 1997). In these guidelines, assumptions are made that 80% of biosolids NH_4-N is lost during application and incorporation, and 15% (anaerobic) and 25% (aerobic) of the biosolids organic N is mineralised in the first year. There is limited information about the applicability of these mineralisation rates of applied biosolids organic N in subtropical Queensland, where it is expected that the mineralisation process could be faster and mineralisation rates higher than those in temperate climate areas. If mineralised N accumulates in the soil profile, it could be lost to the environment by leaching, runoff, and/or denitrification.

Our studies aimed: (*i*) to investigate mineralisation rates of the applied biosolids organic N, as affected by type of biosolids and receiving soil, under subtropical conditions; and (*ii*) to provide better management strategies for beneficial reuse of biosolids in subtropical areas of Australia.

Materials and methods

Sites and biosolids

Four experimental sites were selected, representing major soil types and agricultural cropping systems of subtropical Queensland. All four sites have a history of long, hot summers and mild winters with summer-dominant rainfall.

(1) *Kingaroy site* (28.33S, 152.30E): a Red Ferrosol (Isbell 1996) soil on the Kingaroy Research Station, Queensland Department of Economic Development and Innovation. This is a typical rainfed site in the inland Burnett, growing grain and grain legume crops. Mean annual rainfall is 690 mm.
(2) *Cecil Plains site* (27°40′S, 151°74′E): a heavy Grey Vertosol (Isbell 1996) soil (grey cracking clay) on a private farm. This site is irrigated and is typical of the high-yielding soils of the Darling Downs, growing grain crops and cotton. Mean annual rainfall is 700 mm.
(3) *Bundaberg site* (24°51′S, 152°21′E): a light-textured Yellow Chromosol (Isbell 1996) soil (sandy with increasing clay at depth) on a private farm. This is an irrigated site and grows sugarcane and peanuts. Mean annual rainfall is 1140 mm.
(4) *Lowood site* (27°27′S, 152°34′E): a Black Vertosol (Isbell 1996) soil (black cracking clay) on a private farm in the Brisbane River valley. The site has supplementary irrigation and grew fodder crops. Mean annual rainfall for the nearby town of Lowood is 840 mm.

The three biosolids products used were produced by sewerage wastewater treatment plants from two urban catchments in south-eastern Queensland. One treatment plant produced aerobically digested biosolids sourced from a non-industrial urban area. The second plant produced anaerobically digested biosolids sourced from an urban area with significant industrial activity; these anaerobically digested biosolids were either used immediately after dewatering, or were thermally dried before land application.

Experimental details

Mineralisation trials

There were two mineralisation trials and both were conducted at the Kingaroy site. Trial 1 had eight treatments with three replications and was carried out during dry-season conditions. Treatments comprised an unfertilised control; a fertilised control (commercial fertilisers to supply 65 kg N/ha and 16.5 kg P/ha); two biosolids – aerobically digested, dewatered, and anaerobically digested, dewatered; vermi-composted biosolids; and feedlot manure, with and without composting. The manure and composts were applied at 5 dry t/ha, while the aerobically digested, dewatered biosolids were applied at 14 dry t/ha and the anaerobically digested, dewatered biosolids at 20 dry t/ha. The biosolids application rates were equivalent to an NLBAR calculation designed to supply 180 kg N/ha in the year of application (NSW EPA 1997). The rates of manure and composts were chosen on the basis of typical commercial applications. All materials were broadcast and incorporated by tillage into the top 10–15 cm of soil in microplots with dimensions 1 m by 2 m, with successive plots separated by a 1-m gap. All materials were applied 8 weeks before crop establishment, but biosolids were also applied to separate microplots at 5 and 3 weeks before sowing, with incorporation after each application. Maize (*Zea mays* cv. C79) was planted, and then all plots were irrigated (50 mm) to ensure crop establishment, with a further irrigation (25 mm) applied 1 week later.

Soil samples were collected to depths of 30 cm at 5 and 8 weeks after the initial application, with later samples to a depth of 60 cm at week 15 and 120 cm at week 31. Plants were destructively sampled at week 24 to determine maximum biomass and N uptake, and further samples were collected at week 30 to determine grain yield. (All time periods are relative to the date of the initial biosolids application.)

Trial 2 consisted of eight treatments with four replications in a wetter growing season, with applications of both biosolids and other organic materials calculated to achieve similar rates of total N addition (~500 kg N/ha), although actual rates differed somewhat from that due to differences in N concentrations in the actual batches applied. Treatments comprised an unfertilised control; a fertilised control; and single rates of: aerobically digested, dewatered biosolids; anaerobically digested, dewatered biosolids, planted and unplanted; thermally dried, anaerobically digested biosolids; feedlot manure; and sugarcane mill mud.

Each plot in Trial 2 was 10 m long by 5.4 m wide (equivalent to six crop rows 90 cm apart), with plots separated by 1.8-m-wide laneways. Maize was planted at 7 weeks after biosolids

application. Soil samples were collected to a depth of 90 cm at 11 weeks after application, and collected again to a depth of 150 cm at 31 weeks after application. Plant samples were collected for maximum biomass production and grain harvest at weeks 26 and 31, respectively, relative to the date of biosolids application.

Biosolids rate trials

At each of the four field sites, the general experimental design consisted of the following treatments with three replications: an unfertilised control; a control fertilised with commercial fertilisers applied at recommended rates for the intended crop species; and a series of rates of dewatered biosolids (multiples of NLBAR from 0.5 to 4.5NLBAR) that were broadcast and incorporated before planting. Details of the biosolids application rates, times of application, cropping and rates of artificial fertiliser at each site are summarised in Table 1. In general, soil samples were collected before biosolids application (0–10, 10–20, 20–30, and 30–50 cm) for soil characterisation and after crop harvest (0–10, 10–20, 20–30, and 30–60, 60–90, 90–120, and 120–150/150–180 cm) from the unfertilised control and the NLBAR and 4.5NLBAR treatments. Plant samples were collected at maximum biomass and crop harvest. Both the

Kingaroy and Cecil Plains sites were sampled after the first and second crop cycles and inorganic fertilisers were reapplied after harvesting the first crop.

Calculation of estimated mineralisation rates

The net N of biosolids origin that was assimilated by plants was calculated as the difference in the above-ground crop N between treatments receiving biosolids and the unfertilised control treatments. Such calculations ignore any potential differences in N accumulation in plant root systems, but as root mass accounts for a relatively small proportion of the whole plant biomass, we have assumed that differences in root N accumulation would be minimal.

Organic soil N and inorganic soil N were quantified by measuring concentrations in the 0–30 cm and 0–150/180 cm of the soil profile, respectively, and converting concentrations to quantities per hectare using measured bulk densities in each profile layer. The organic and inorganic N that had been derived from biosolids applications was assessed by subtracting the organic N and NO_3-N in each layer of the unfertilised control profile from those in the respective layers of the different biosolids treatments. Profile totals were calculated by summing the measured profile layers. Partial N budgets

Table 1. Biosolids rate trials: field experimental details and important dates at each site
Application rates of biosolids are multiples of NLBAR from 0.5 to 4.5NLBAR

Treatment	Application rates	Application date	Crop	Planting date	Soil sample date	Plant harvest date
		Kingaroy, Red Ferrosol				
Anaerobic biosolids	12.5, 25, 37.5, 50, 75, 112.5 dry t/ha	24/01/02	Millet Maize	7/02/02 8/02/03	08/05/02 24/03/03 10/09/03	07/05/02 09/09/03
Aerobic biosolids	5, 10, 15, 20, 30, 45 dry t/ha	24/01/02				
Fertilised control	0.180 t N/ha, 0.034 t P/ha	07/02/03				
		Bundaberg, Yellow Chromosol				
Anaerobic biosolids	10, 20, 30, 40, 60, 90 dry t/ha	11/04/02	Sugarcane	15/04/02	17/10/02	16/07/02 16/09/03
Aerobic biosolids	7, 14, 21, 28, 42 dry t/ha	11/04/02				
Fertilised control	0.173 t N/ha, 0.064 t P/ha	11/04/02				
		Cecil Plains, Black Vertosol				
Anaerobic biosolids	10, 20, 30, 40, 60, 90 dry t/ha	11/09/02	Sorghum Cotton	23/10/02 18/10/03	06/03/03 26/11/03 19/05/04	04/03/03 05/05/04
Aerobic biosolids	7, 14, 21, 28, 42 dry t/ha	11/09/02				
Fertilised control	0.180 t N/ha, 0.034 t P/ha	23/10/03				
		Lowood, Black Vertosol				
Anaerobic biosolids	10, 20, 30, 40, 60, 90 dry t/ha	15/10/02	Forage sorghum	23/10/02	30/05/03	01/12/02 07/01/03 11/03/03
Aerobic biosolids	7, 14, 21, 28, 42 dry t/ha	15/10/02				
Fertilised control	0.180 t N/ha, 0.045 t P/ha	15/10/02				

were calculated for the various biosolids treatments by subtracting the net N in above-ground plant biomass and the soil organic and NO_3-N pools from the biosolids N added. The biosolids N that could not be accounted for in these calculations was termed 'lost N'. The fate of this N was not determined in this study, but it was assumed to have moved off-site due to a combination of gaseous N loss pathways (ammonia volatilisation and/or denitrification) and leaching losses, with the latter evidenced by enriched NO_3-N concentrations in the deepest profile layers compared with the unfertilised control. The only instance in which leaching had obviously occurred was at the sugarcane site in the Yellow Chromosol at Bundaberg (Barry *et al.* 2006).

The estimated N mineralised (%) from biosolids organic N was calculated as: $100 \times$ (net mineral N of biosolids origin removed by plant + net soil profile NO_3-N of biosolids origin + N lost – NH_4-N applied in biosolids – NO_3-N applied in biosolids)/biosolids organic N, assuming that all the N lost and N removed by the plant was in the mineral N form.

Chemical analyses

Soil and plant samples collected from the field were oven-dried at 40°C and 60°C, respectively, and then ground <2 mm before chemical analysis.

Standard soil fertility analyses were conducted using methodology outlined in Rayment and Higginson (1992). The pH and electrical conductivity of both soil and biosolids were determined using a 1 : 5 soil/biosolids to water ratio. Soil and biosolids total potassium (K), P, sulfur (S), cadmium (Cd), Cu, magnesium (Mg), manganese (Mn), and Zn were determined by inductively coupled plasma-atomic emission spectroscopy following digestion using a concentrated acid mixture of 3 : 1 HNO_3 : HCl (by volume).

Biosolids and soil profile mineral N (NH_4-N and NO_3-N) contents were extracted using a 1 : 10 ratio of biosolids/soil : 2 M KCl solution, and then determined by an automated colourimetric method (Method 7C2, Rayment and Higginson 1992). The quantity of mineral N determined by this method included both NO_2-N and NO_3-N. Biosolids and soil total N (0–30 cm) contents were determined by a semi-micro Kjeldahl digestion method. Plant total N and carbon (C) and soil total C were determined by a dry-combustion method using a LECO CNS-2000 analyser (LECO Corporation, MI, USA).

Statistical analyses

Differences in each N pool were compared between unfertilised control and biosolids/organic amendment treatments using standard analysis of variance techniques. Differences between the treatment means were tested for statistical significance using the least significant difference (l.s.d.) procedure in the GENSTAT® (6th Edn) statistical package. Profile total N (0–30 cm) and mineral N (150/180 cm) were calculated by summation of individual depths.

Results and discussion

Characterisation of soils and biosolids

The soils at the four field trial sites differed in their physical and chemical characteristics (Table 2), and were representative of some of the major cropping soils in proximity to major urban biosolids production facilities in south-eastern Queensland. The soils ranged from alkaline heavy clays (Cecil Plains) to neutral flood-plain alluvial clay loams (Lowood) and mildly acidic, lighter textured clays (Kingaroy) or sandy loams (Bundaberg).

Soil organic C ranging between 1.3% and 1.8% at Kingaroy, Bundaberg, and Cecil Plains is typical of conventionally cropped soils in southern Queensland, while the much higher soil organic C at Lowood (4.4%) was associated with a preceding pasture phase on a dairy farm. Most soil N was in the organic form at all sites, with little mineral N in the topsoil (0–10 cm) when the study began, at the Lowood and Bundaberg sites in particular. The high organic C concentration (4.4%) and the relatively low C : N ratio (11 : 1) at the Lowood site suggested significant reserves of potentially mineralisable N for subsequent crop use.

Potential N loss pathways varied significantly between locations. The well-drained Red Ferrosols at Kingaroy (Bell *et al.* 2005) have the potential to experience significant leaching losses of NO_3-N mineralised from applied biosolids, but risks were reduced by a relatively dry rainfed production environment (average annual rainfall 780 mm). Risks were much greater at Bundaberg, where the coincidence of a production system employing overhead irrigation, higher average annual rainfall (1070 mm), and a sandy soil (69% sand) raised significant concerns about leaching losses. Conversely, while the heavier clay soils at Lowood and Cecil Plains suggested lower rates of internal drainage and hence leaching potential, risks of gaseous N losses were increased. The high soil pH at Cecil Plains suggested a potential risk of N loss through ammonia volatilisation, while reduced internal drainage rates, flood irrigation (Cecil Plains), and high background soil organic C (Lowood) or high crop residue inputs in a high-yielding production system (Cecil Plains) suggested potential for denitrification if mineralisation of biosolids N exceeded crop demand.

Table 2.   Selected soil (0–10 cm) properties of the experimental sites
Values are the means of three replicates. n.d., Not detected

Site	Soil type	pH[A]	EC (dS/m)	NH_4-N	NO_3-N (mg/kg)	Colwell P	K_{total}	P_{total}	S_{total}	C_{total} (%)	N_{total}	Sand	Silt	Clay	C : N ratio
Kingaroy	Red Ferrosol	5.9	0.09	n.d.	22	56	0.090	0.053	0.03	1.8	0.20	22	19	60	9
Cecil Plains	Grey Vertosol	8.1	0.12	5	15	20	0.43	0.03	0.01	1.3	0.09	14	15	71	14
Bundaberg	Yellow Chromosol	6.2	0.05	n.d.	1	41	0.29	0.03	<0.01	1.6	0.08	69	16	15	20
Lowood	Black Vertosol	6.8	0.22	n.d.	<1	320	0.34	0.15	0.05	4.4	0.40	22	21	57	11

[A]1 : 5 H_2O.

Table 3. Selected properties of biosolids and organic amendments before application

TKN, Total Kjeldahl N. Values are the means of three replicates and based on oven-dried weight at 105°C

Product	pH [A]	NH$_4$-N	NO$_3$-N	Moisture	C$_{total}$	TKN	P$_{total}$	K$_{total}$	S$_{total}$	C : N
		(mg/kg)				(%)				ratio
Aerobic dewatered biosolids	6.5	2612	10	88	32	6.1	4.3	0.9	0.8	5.2
Anaerobic dewatered biosolids	7.6	12 690	14	80	34	6.1	2.8	0.3	1.5	5.6
Thermally dried, anaerobic dewatered biosolids	7.0	4540	4	9	36	5.9	2.4	0.3	1.7	6.9
Vermicasts derived from biosolids	6.1	180	32	17	12	1.4	1.9	0.3	0.6	8.6
Feedlot manure	7.9	835	80	16	42	3.3	0.6	2.1	0.6	12.7
Composted feedlot manure	7.6	na	na	28	21	2.6	1.1	2.5	0.7	8.1
Sugarcane mill mud	6.8	10	6	78	31	1.4	1.0	0.6	0.15	22.1

[A]1 : 5 H$_2$O.

The dewatered biosolids represented a rich source of N, P, and S (Table 3), with higher concentrations of all nutrients than in any of the other organic amendments. About 20% of the total N in the anaerobic dewatered biosolids was present as NH$_4$-N, but this percentage was much reduced in the same biosolids after a thermal drying process (8.7% of total N) and was even lower (5%) for the aerobic dewatered biosolids, lower again for the manure (2.5%), and almost non-existent for the other materials. The high NH$_4$-N concentration of the anaerobic biosolids, combined with its neutral–slightly alkaline pH, predisposes this type of amendment to gaseous N losses due to NH$_3$ volatilisation (Robinson and Röper 2003). Furthermore, the high N concentrations and very low C : N ratios (5–7 : 1) of all of the biosolids suggests the potential for rapid N mineralisation. The resulting accumulation of NO$_3$-N in the soil or in biosolids lumps, when combined with wet soil conditions, could provide conditions conducive to denitrification. These risks are amplified by the addition of a polymer to enhance flocculation during biosolids dewatering, with this polymer enhancing moisture retention in biosolids lumps and increasing denitrification losses (Pu *et al.* 2010).

Mineralisation trial 1

During the first 8 weeks after biosolids incorporation, the dominant component of the soil mineral N pool for both biosolids treatments was NH$_4$-N, representing 65–80% (anaerobic) or 55–65% (aerobic) of the total mineral N in the top 30 cm of the soil profile (Table 4). This compares with 30% for the manure and 8–12% for the unfertilised control, vermicompost, and composted manure amendments during the same period. The effect was most obvious in the top 10 cm of the soil profile, corresponding to the depth of incorporation of the amendments, but was also transitory, as NO$_3$-N had become the dominant form of mineral N in the same profile layers in later samplings at 15 weeks (75% of total mineral N) and 31 weeks (90–95% of total mineral N) after biosolids application (data not presented).

This NH$_4$-N dominance in the biosolids treatments was unexpected. While the anaerobic biosolids contained significant amounts of NH$_4$-N at the time of application (24% of the total N was as NH$_4$-N and only 0.03% as NO$_3$-N), current guidelines in the NLBAR calculations derived from the NSW studies suggest 80% of that NH$_4$-N will be lost due to volatilisation of NH$_3$ during spreading and incorporation (NSW EPA 1997). The aerobic biosolids contained very little mineral N (~4.7% of the total N as NH$_4$-N and only 0.03% as NO$_3$-N). It therefore seems likely that the rapid accumulation of mineral N, and the early dominance of NH$_4$-N over NO$_3$-N in the soil mineral N pools, was due to *in situ* mineralisation of organic N from the biosolids. Soil conditions were quite dry for most of

Table 4. Mineralisation trial 1. Effects of application of biosolids and manures on concentrations of NH$_4$-N and NO$_3$-N in the 0–10, 10–20, and 20–30 cm soil layers, total mineral N in the top 30 cm of the soil profile, and the percentage of that mineral N present as NH$_4$-N

All data except NH$_4$-N% are transformed data (log(x+1)), with back-transformed means shown in parentheses. n.s., Not significant

Treatment	Weeks after addition	0–10 cm		10–20 cm		20–30 cm		Tot. mineral N (0–30 cm)	NH$_4$-N
		NH$_4$-N	NO$_3$-N	NH$_4$-N	NO$_3$-N	NH$_4$-N	NO$_3$-N	(kg N/ha)	(%)
		(mg/kg)		(mg/kg)		(mg/kg)			
Control	0	0.742 (4.5)	1.276 (17.9)	0.189 (0.5)	1.314 (19.6)	0.193 (0.6)	1.326 (20.2)	1.864 (72.1)	8.3
	3	0.952 (8.0)	1.509 (31.3)	0.468 (1.9)	1.495 (30.3)	0.541 (2.5)	1.465 (28.2)	2.070 (116.5)	12.0
	8	0.936 (7.6)	1.577 (36.8)	0.239 (0.7)	1.542 (33.8)	0.464 (1.9)	1.505 (31.0)	2.108 (127.2)	9.1
Anaerobic biosolids	3	2.178 (149.7)	1.476 (28.9)	1.149 (13.1)	1.465 (28.2)	1.089 (11.3)	1.395 (23.8)	2.461 (288.1)	65.2
	5	2.495 (311.6)	1.698 (48.9)	1.690 (48.0)	1.598 (38.6)	1.610 (39.7)	1.538 (33.5)	2.792 (618.4)	76.8
	8	2.514 (325.6)	1.679 (46.8)	1.584 (37.4)	1.512 (31.5)	1.736 (53.5)	1.536 (33.4)	2.808 (641.7)	79.7
Aerobic biosolids	3	2.052 (111.7)	1.608 (39.6)	1.332 (20.5)	1.539 (33.6)	1.193 (14.6)	1.389 (23.5)	2.446 (278.3)	60.4
	5	1.085 (11.2)	1.476 (28.9)	1.131 (12.5)	1.450 (27.2)	2.200 (157.5)	1.659 (44.6)	2.559 (361.2)	64.5
	8	1.509 (31.3)	1.444 (26.8)	1.528 (32.7)	1.638 (42.5)	1.889 (76.4)	1.702 (49.4)	2.613 (409.2)	55.1
Vermicast biosolids	8	0.980 (8.5)	1.607 (39.5)	0.395 (1.5)	1.393 (23.7)	0.414 (1.6)	1.437 (26.4)	2.064 (114.9)	11.1
Feedlot manure	8	1.436 (26.3)	1.580 (37.0)	0.729 (4.4)	1.485 (29.5)	0.699 (4.0)	1.417 (25.1)	2.177 (149.3)	29.6
Composted manure	8	0.813 (5.5)	1.548 (34.3)	0.813 (1.3)	1.389 (23.5)	0.450 (1.8)	1.375 (22.7)	2.013 (102.0)	9.9
l.s.d. (*P* = 0.05)		0.539	0.155	0.623	n.s.	0.665	n.s.	0.320	18.9

the period from the initial incorporation of treatments in late November until the irrigations applied at planting on 21 January. In total, only 40 mm of rain fell in the period, of which 28 mm fell over 2 days in the fourth week after the initial application. The relatively dry soil conditions might have slowed nitrification of the accumulating NH_4-N, as this process is known to be more sensitive to moisture deficit than ammonification (Stark and Firestone 1995).

The rate at which mineral N accumulated in the biosolids treatments during the 8 weeks before maize planting was extremely rapid (Table 4). During the 8 weeks from initial incorporation of amendments, background N mineralisation in the unfertilised control treatment released 50–55 kg N/ha in the top 30 cm of the profile. Assuming a similar starting mineral N content at time 0 for all amendments, there were no significant increases in total mineral N, compared with the unfertilised control, in the raw manure, vermi-compost, or composted manure treatments after 8 weeks. However, both biosolids treatments already showed significantly greater mineral N than the control treatment 3 weeks after incorporation, with increases continuing throughout the following 5 weeks. After 8 weeks, the soil mineral N content in the anaerobic biosolids treatment had increased by 514 kg N/ha, while that in the aerobic biosolids treatment had increased by 280 kg N/ha (Table 4). These increases represented 40% and 33% of the total applied N for the anaerobic and aerobic biosolids treatments, respectively. In the case of the aerobic biosolids, this increase would have necessitated mineralisation of 58% of the added organic N, given the small amount of mineral N in this material. However, the situation is less certain in the anaerobic biosolids, where ~20% of the biosolids N was added as NH_4-N. Our calculation of 40% of the added N in the soil mineral N pool assumes conservation of this NH_4-N, but if the assumption of 80% losses of NH_4-N during spreading and incorporation applies (NSW EPA 1997) and this N never enters the soil N pool, the net increase in soil mineral N actually represents 46% of the added N. The implications for mineralisation of organic N are even greater, with the mineralisation of organic N necessary to increase the soil mineral N pool by the observed 514 kg N/ha varying from 28% (NH_4-N conservation) to 44% (80% loss of NH_4-N) of the added organic N.

A midseason soil mineral N assessment taken 22 weeks after the initial incorporation (data not shown) showed similar net mineral N increases in the top 50 cm of the profile of unplanted anaerobic and aerobic biosolids treatments relative to the unfertilised control (i.e. an additional 570 and 290 kg mineral N/ha, respectively), while totals in treatments sown to maize were lower, presumably due to plant uptake. However, all biosolids treatments showed significantly greater mineral N concentrations in the deepest soil layer sampled (30–50 cm) than the respective planted or unplanted controls, indicating leaching of mineral N to deeper soil layers. There were no such differences between the other organic amendments and the control treatments in this layer (data not presented).

The final profile sampling at harvest of the maize crop 31 weeks after the initial incorporation showed no evidence of significant leaching of mineral N below the depth of sampling, even in the unplanted anaerobic and aerobic biosolids treatments (Fig. 1*a*). This was not surprising, given that the total rainfall during the 31-week period was only 360 mm. The absence of leaching enabled a comparative assessment of soil mineral N contents between all treatments, and by including N in above-ground plant biomass, allowed a quantification of the net mineral N contribution of the various amendments during the growing season (Fig. 2). The high N-mineralisation capacity of the Red Ferrosol soil on which this study was conducted is illustrated by the fact that plants in the unfertilised control treatment were able to accumulate 127 kg N/ha, mostly from the top 60 cm of the soil profile (Fig. 1*a*), in dry seasonal conditions that were not conducive to deep profile wetting and root proliferation. Approximately 65% of the residual mineral N in the soil profile at harvest in the unfertilised control treatment (218 kg N/ha) was at depths >60 cm.

Only the anaerobic and aerobic biosolids treatments were able to significantly increase N accumulation in maize plant biomass, with 52–64 kg N/ha accumulated in the anaerobic biosolids treatments and 59–69 kg N/ha in the aerobic biosolids treatments (Fig. 2). These net N contributions were significantly greater than those of the vermi-compost and raw or composted manures, which made a net contribution of 24–26 kg N/ha. Treatment differences were even greater for soil mineral N, with the greatest net contributions from anaerobic biosolids (180–230 kg N/ha), followed by aerobic biosolids (80–150 kg N/ha). Except for the latest applied aerobic biosolids treatment (3 weeks before planting), all provided significant increases in profile mineral N and were not different from each another. The contrast with the other amendments was strong, with none of those treatments making any net mineral N contribution at the time of sampling (–30 to +12 kg N/ha, relative to the unfertilised control).

Interestingly, the combined contribution of amendments to mineral N recovered at the end of the season (sum of soil and plant N, assuming plant N was accumulated as mineral N) ranged from 240–270 kg N/ha in the anaerobic biosolids treatment to 150–210 kg N/ha in the aerobic biosolids treatment. This was much less than the mineral N recorded in the top 30 cm of the soil profile at planting (8 weeks after the initial biosolids applications, Table 4), when the net mineral N contributions were 514 and 280 kg N/ha for the anaerobic and aerobic biosolids treatments, respectively. A small amount of this 'missing' mineral N may have been incorporated in plant root systems, which were not quantified in this study, although given the relatively small net N contribution to above-ground N contents (Fig. 2), any additional N in roots was likely to be minimal. Some of this N may also have been re-immobilised in the soil organic N pool (component discussed later in the context of partial N budgets).

Mineralisation trial 2

In contrast to trial 1 when the period after incorporation was hot and dry, trial 2 experienced consistently moist seasonal conditions. During the 11 weeks from incorporation until the first soil mineral N assessment at 3 weeks after maize planting, 116 mm of rainfall was recorded, with a total rainfall of 634 mm during the 31-week experimental period. The mineral N content in the top 30 cm of the soil profile measured at 11 weeks after incorporation of the various amendments, and the proportion of
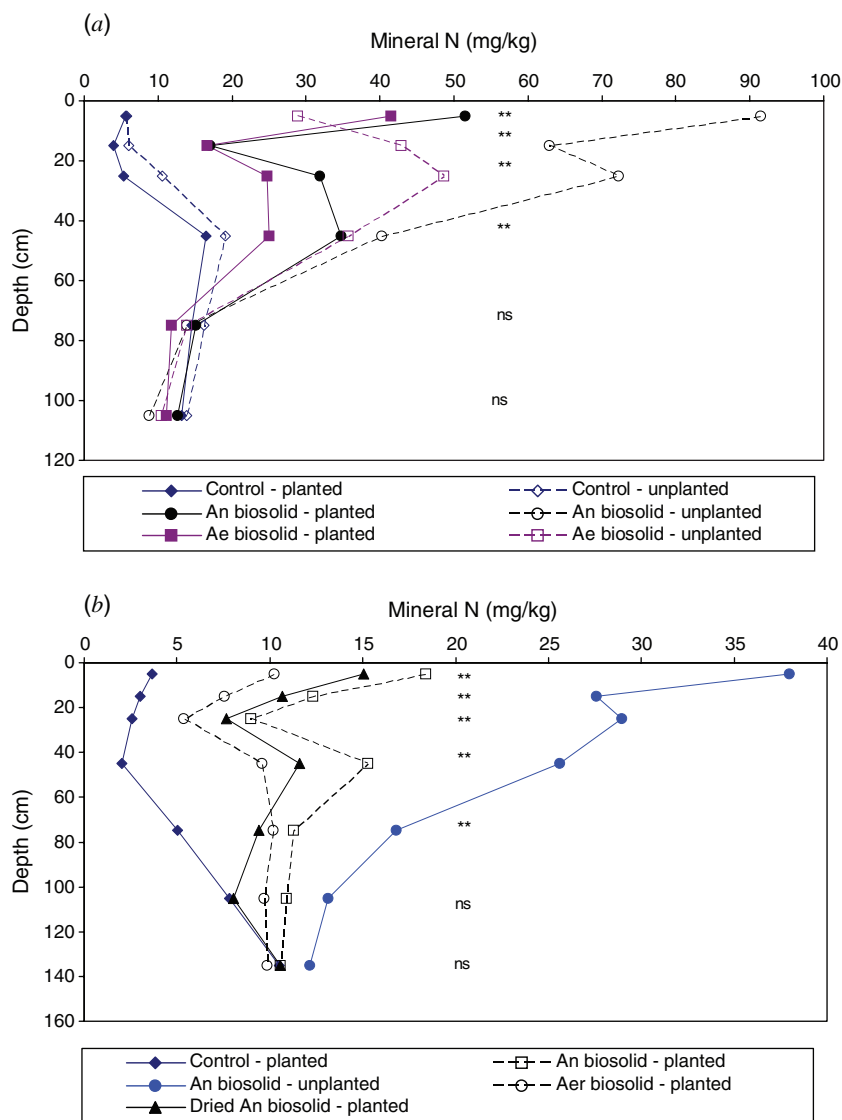
Fig. 1. Transformed ($\log(x+1)$) soil mineral N profiles (NH_4-N + NO_3-N, mg N/kg) for the unfertilised controls and the anaerobic (An) and aerobic (Ae/Aer) biosolids treatments at the end of the crop season at 31 weeks after incorporation for (*a*) mineralisation trial 1 and (*b*) mineralisation trial 2. Data in (*a*) represent means of the three biosolids application times.

that mineral N present as NH_4-N, are shown in Table 5, with data from trial 1 at 8 weeks after incorporation shown for comparison. Soil mineral N data from trial 2 would have had little effect from the establishing maize plants, given that samples were taken only 3 weeks after sowing.

Two differences between the data from trials 1 and 2 are apparent. The first difference was the lesser amount of mineral N in the same top 30 cm soil layer in trial 2. Part of this lower mineral N was associated with the lower background mineral N in the control soil, with 24 kg mineral N/ha in the top 30 cm, compared with 127 kg mineral N/ha in trial 1 (Table 5). However, even allowing for the lower background mineral N in trial 2, mineral N contents were low. This was particularly evident in the manure treatment, applied at more than five times the N application rate of trial 1 and yet producing similar

amounts of net mineral N. Similarly, while the N application rates in the biosolids treatments were only 40% (anaerobic dewatered biosolid) to 60% (aerobic dewatered biosolid) of those used in trial 1, net mineral N contribution was only 14% (anaerobic) to 23% (aerobic) of that in trial 1 in the top 30 cm of the profile. There were slight increases in the mineral N contents in the 30–60 cm layer in trial 2, which were consistent with some leaching of NO_3-N, but this amounted to <10 kg mineral N/ha (data not shown) and could not account for the observed differences.

The second difference between the two mineralisation experiments was in the relative proportions of NH_4-N and NO_3-N, especially in the biosolids treatments (Table 5). Trial 1 exhibited much higher proportions of NH_4-N in the rapidly mineralising biosolids treatments, with an average of
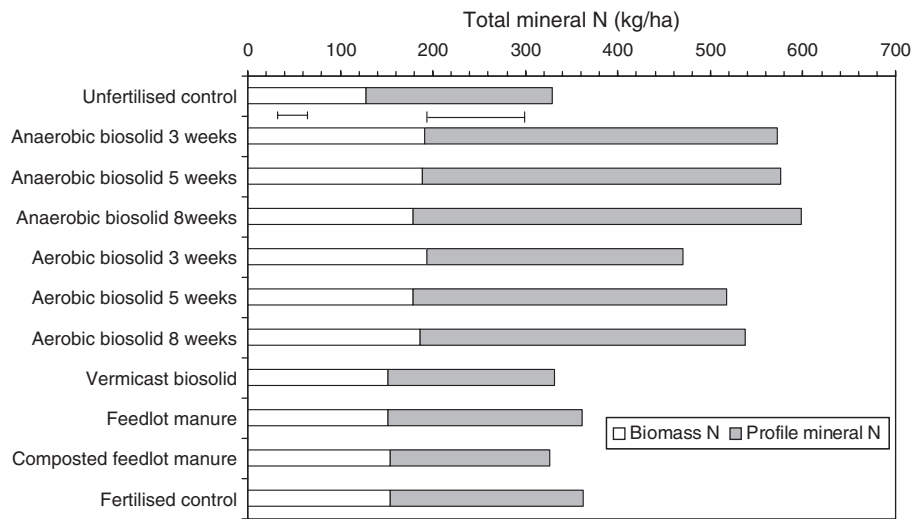
Fig. 2. Mineral N present in soil or above-ground maize biomass at the end of mineralisation trial 1 in 2004. All organic amendments were incorporated at 8 weeks before planting, except the anaerobic and aerobic biosolids treatments applied 3 weeks and 5 weeks before planting. Horizontal bars indicate l.s.d. ($P=0.05$) values for plant biomass N (38 kg N/ha) and profile mineral N (105 kg N/ha); l.s.d. ($P=0.05$) for combined total mineral N (plant + soil) is 117 kg N/ha.

Table 5.   Mineralisation trials 1 (2003) and 2 (2004): Comparison of mineral N content in the top 30 cm of the soil profile (kg/ha) and the percentage of that mineral N present as NH₄-N

Values for mineral N are log with transformed data ($\log(x+1)$) in parentheses. Sampling times were 8–11 weeks after incorporation of organic amendments

| | Profile mineral N to 30 cm (kg/ha) | | | Mineral N present as NH₄ (%) | | |
| | Trial 1 (8 weeks after incorporation) | Trial 2 (11 weeks after incorporation) | | Trial 1 (8 weeks after incorporation) | Trial 2 (11 weeks after incorporation) | |
	Unplanted	Planted	Unplanted	Unplanted	Planted	Unplanted
Unfertilised control	2.108 (127.2)	1.39 (23.5)		9.1	28.9	
Anaerobic dewatered biosolids	2.808 (641.7)	1.933 (84.7)	2.099 (124.6)	79.7	23.3	19.3
Thermally dried anaerobic dewatered biosolids		1.981 (94.7)			16.3	
Aerobic dewatered biosolids	2.613 (409.2)	1.951 (88.3)		55.1	23.2	
Vermicast biosolids	2.209 (114.9)			11.1		
Feedlot manure	2.177 (149.3)	1.764 (57.1)		29.6	20.1	
Composted feedlot manure	2.013 (102.0)			9.9		
Sugarcane mill mud		1.565 (61.4)			31.3	
l.s.d. ($P=0.05$)	0.32	0.101		18.9	9.1	

55% (aerobic) to 80% (anaerobic) NH₄-N, compared with only 16–23% NH₄-N in trial 2. Trial 1 also showed significant differences in the proportion of mineral N present as NH₄-N between amendments (biosolids >> feedlot manure > composts and unfertilised control soil), whereas in trial 2, the ranking was effectively reversed (i.e. mill mud and unfertilised control soil ≥manure and biosolids treatments). The reduced presence of NH₄-N in trial 2 was likely to be a function of the wetter soil moisture conditions, which would have resulted in rapid conversion of NH₄-N to NO₃-N (Stark and Firestone 1995).

Similar to mineralisation trial 1, the final profile sampling at the time of harvest of the maize crop at 31 weeks after treatment incorporation showed no evidence of significant leaching losses of mineral N below the depth of sampling, even in the unplanted anaerobic biosolids treatments (Fig. 1*b*). However, consistent with the wetter seasonal conditions, there were differences in

mineral N contents deeper into the soil profile, with significant increases in the 60–90 cm layer and suggestions of slight increases in the 90–120 cm layer below. The collection of soil samples below the leaching zone again enabled a comparative assessment of the net contribution of the various amendments to mineral N retained in the profile or accumulated in plant biomass at the end of the growing season (Fig. 3). All amendments resulted in significant increases in plant N uptake relative to the unfertilised control, with the biosolids (44–61 kg N/ha net increase) again making slightly greater contributions than the other amendments (32–37 kg N/ha). The increase in soil mineral N was much greater than that of plant N uptake, especially for the biosolids treatments. Profile mineral N increased by 263 kg N/ha in the anaerobic biosolids treatment, by 172 (dried) and 160 (aerobic) kg N/ha in the other biosolids treatments, but by only 88 kg N/ha in the mill mud and

Total mineral N (kg/ha)
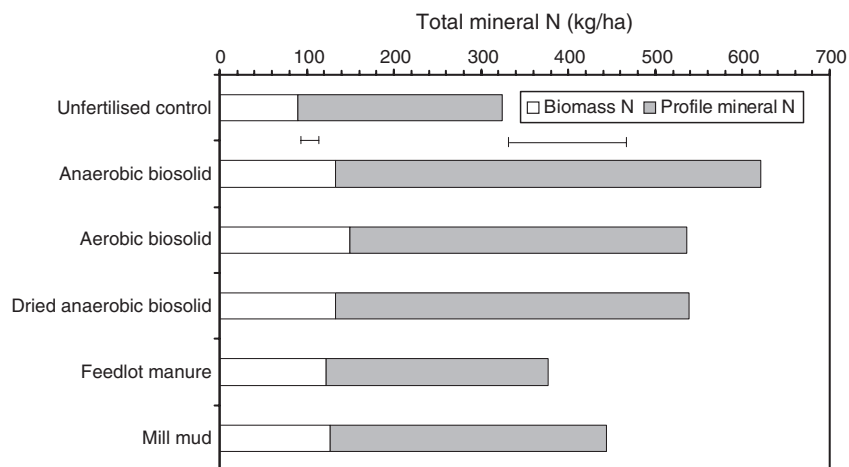
Fig. 3. Mineral N present in soil or above-ground maize biomass at the end of mineralisation trial 2 in 2005. Horizontal bars indicate l.s.d. ($P = 0.05$) values for plant biomass N (20 kg N/ha) and profile mineral N (142 kg N/ha); l.s.d. ($P = 0.05$) for combined total mineral N (plant + soil) is 149 kg N/ha.

31 kg N/ha in the manure treatments. Collectively, the net contributions to plant and soil N in the different treatments therefore ranged from 63 kg N/ha in the manure treatment to 308 kg N/ha in the anaerobic biosolids treatment.

We compare these totals with those recorded for similar materials that were used in mineralisation trial 1 (see Fig. 2). Despite the reduced rates of N addition in the anaerobic and aerobic biosolids in trial 2 (i.e. 672 *v.* 1276 kg N/ha for anaerobic biosolids and 513 *v.* 855 kg N/ha for aerobic biosolids) and the greatly increased rates of N addition in the manure (810 *v.* 146 kg N/ha), totals recovered in soil mineral N and plant N were not greatly different between trials. Totals in trial 1 were 253, 179, and 36 kg N/ha for the anaerobic and aerobic biosolids and feedlot manure, respectively, compared with 308, 221, and 63 kg N/ha for the same materials in trial 2. The manure provided a much lower 8% of the total N added to the collective soil mineral N and plant N pools, compared with 22% in trial 1 over a similar 31-week assessment period, and this was consistent with an expectation of greater immobilisation in soil organic N pools in the wetter seasonal conditions. However, both biosolids appeared to provide much greater proportional contributions when applied at the lower rates in trial 2 (46% *v.* 20% for the anaerobic biosolids and 32% *v.* 21% for the aerobic biosolids). These differences were consistent with accelerated rates of mineralisation of organic N in the wetter, second season, but the retention of mineralised N in inorganic forms in these treatments was not consistent with the greater potential for subsequent re-immobilisation. Indeed, these higher mineralisation rates and the retention of that N as mineral N in the soil would predispose such treatments to gaseous N losses.

Data on soil organic N pools in the top 30 cm of the profile at the end of each experimental period were analysed in an attempt to detect residual applied organic N and/or re-immobilised mineral N in the soil profile (Table 6). These analyses have limitations associated with the detection limits for soil total N (0.01%, or ~260 kg N in the top 30 cm of the soil profile), but where rates of N addition were substantial, there was an opportunity to use the data, in combination with profile

mineral N and plant N measurements, to construct partial N budgets for each study (Table 6) and estimate gaseous N losses. A major uncertainty with this approach was how to account for the large amounts of NH_4-N in the anaerobic biosolids, as the current NLBAR calculation assumes 80% of this NH_4-N will be lost during spreading and incorporation (NSW EPA 1997). In our calculations we have assumed conservation of this added NH_4-N, so part of the lost N (i.e. unable to be accounted for in these partial N budgets) in these calculations for the anaerobic biosolids treatments could represent the expected volatilisation loss from the added NH_4-N.

Calculated net N-mineralisation rates of organic N in the biosolids treatments in both trials were significantly greater than the 15% (anaerobic dewatered biosolids) to 25% (aerobic dewatered biosolids) suggested under the current guidelines for determining NLBAR application rates (NSW EPA 1997). In trial 1, when the NLBAR calculation was used to determine the rate of biosolids N addition, the high mineralisation rates were driven in large part by the substantial amounts of assumed 'lost' N (350–450 kg N/ha), which was not able to be detected in plant biomass or the soil mineral and organic N pools. This N was assumed to have been lost by either volatilisation of NH_3 or denitrification of N_2 or N_2O. The very high NH_4-N concentrations recorded in the top 10–20 cm of the soil profile during the early dry conditions in trial 1 (see Table 4) would have increased the chances of volatilisation losses. Subsequent wetting events would have facilitated rapid nitrification, but the enhanced moisture retention in biosolids lumps due to the presence of polymers used in biosolids dewatering, combined with a lack of major leaching rainfall events, would then have allowed development of conditions favourable to denitrification losses. Indeed, Pu *et al.* (2010) have shown that denitrification losses can be a major loss pathway for biosolids N, with losses accentuated by the presence of the flocculent polymer added during biosolids dewatering.

The amounts and proportions of 'lost N' in the biosolids treatments appeared to be reduced in trial 2, when lower

Table 6.   Mineralisation trials: Estimated mineralisation rates of applied organic N in biosolids and organic amendments calculated at 31 weeks after application to a Red Ferrosol soil in either a dry (trial 1) or wet (trial 2) summer growing season

l.s.d. (*P* = 0.05) values are shown for the net recovered N in plant biomass or soil organic or mineral N pools. n.r., Not reported due to the small total amounts of N added in the vermicast, manure, and composted manure products relative to the errors associated with measurement of total soil N; n.s., not significant

	Biosolids type	Applied N (kg/ha)			Net N recovered (kg/ha)				Lost N (kg/ha)	Tot. mineral N including losses[C] (kg/ha)	Net mineralised N[D]	
		Total	Mineral	Organic[A]	Plant	Mineral	Organic[B]	Total			(kg/ha)	% Organic N applied
Trial 1	Anaerobic	1276	212	1064	59	194	669	922	354	607	395	37
	Aerobic	855	35	820	59	120	220	399	456	635	600	68
	Vermicasts	70	<1	69	25	−29	n.r.	n.r.	n.r.	−4[E]	−4[E]	0[E]
	Manure	146	4	142	24	12	n.r.	n.r.	n.r.	36[E]	32[E]	22[E]
	Compost	112	3	109	27	−29	n.r.	n.r.	n.r.	−2[E]	−5[E]	0[F]
	l.s.d. (*P* = 0.05)				36	115	292	285	n.s.			29
Trial 2	Anaerobic	672	139	534	45	263	356	664	8	316	178	33
	Dried anaerobic	513	21	492	44	172	282	498	15	231	210	43
	Aerobic	691	36	655	61	160	345	567	124	346	310	47
	Manure	810	29	781	32	31	442	505	305	368	339	43
	Mill mud	346	<1	345	37	88	187	312	34	159	158	46
	l.s.d. (*P* = 0.05)				21	37	n.s.	n.s.	n.s.			n.s.

[A]Calculated as the difference of (total N – mineral N).

[B]Calculated as the difference of (total N – mineral N – plant N).

[C]Calculated as the sum of (net soil mineral N, net plant N, and lost N at the end of the experimental period). This assumes that 'lost N' had to have undergone prior mineralisation.

[D]Calculated as (net total mineral N at the end of the study – mineral N added in amendments), expressed as kg N/ha or as a % of added organic N. This assumes total conservation of N added as NH_4-N.

[E]Minimum values derived from net soil and plant N only.

application rates were used. However, these lower addition rates, combined with larger plots and the use of mechanical spreading equipment (compared with the use of small microplots and careful hand spreading in trial 1), introduced far greater errors in the measurement of soil organic N and hence uncertainty in estimates of lost N. This resulted in considerable variability between replicates and a lack of significant differences between treatments in net organic N recovered. Even with these limitations, however, data again suggest greater rates of organic N mineralisation than currently used in the NLBAR calculations (NSW EPA 1997).

Results for other organic amendments were inconsistent. Trial 1 results suggested small amounts of N recovery in plant biomass and negligible residual mineral N in the soil profile (Fig. 3). Rates of N addition were too small to attempt detection of residual organic N, but assuming gaseous losses were minimal (mineral N concentrations were not high enough to facilitate losses by either pathway, Table 4), the only material to mineralise a significant amount of organic N was the feedlot manure (22% of the organic N added). Trial 2, although receiving larger amounts of N, suffered variability issues similar to those discussed for the biosolids treatments. The net N contributed to the plant and mineral N pools was much less than with the biosolids, with the sugarcane mill mud producing proportionally much greater net plant and soil mineral N returns than the manure (i.e. ~30% *v.* 8% of total N added). The greatest net contribution to the soil organic N pool was in the manure treatment, consistent with the greatest rate of N addition (Table 6). However, like the biosolids, both the manure and mill mud seemed to retain ~50–55% of the total added N in the organic N pool. The noticeably high amount of lost N for the manure treatment (~35% of the total added N) was surprising,

and was inconsistent with the low net residual mineral N and plant N contributions. As such, this result may be due (at least in part) to the variability in spreading and then measuring soil total N, rather than an apparently very rapid mineralisation of >40% of the added organic N in that amendment.

Biosolids rate trials

Data from the field trials indicated that plant growth and yields in all biosolids treatments over the assessment period considered (i.e. 3.5–18 months, or 1–2 crop seasons) were not limited by availability of N, and that significant differences in N uptake occurred in response to biosolids applications. However, net plant N uptake in biosolids treatments accounted for only a small amount of applied biosolids N at all sites except Lowood (Fig. 4), especially for the 4.5NLBAR treatments. At the Bundaberg, Kingaroy, and Cecil Plain sites, the net amount of N taken up by the plant (including N removed in grain of the first crop over the longer assessment periods at Kingaroy and Cecil Plains) ranged from 23 to 150 kg N/ha for 1NLBAR treatments, i.e. 3–15% of the total N applied. The 4.5NLBAR rates resulted in only slightly greater net plant N uptake (50–196 kg N/ha), representing <1–5% of the total N added. However, under the more exploitive cut-and-remove forage system at the Lowood site, much greater net N removal was recorded: 244 and 248 kg N/ha for 1NLBAR treatments (25–33% of applied N) and 365–488 kg N/ha (8–15% of applied N) for the 4.5NLBAR treatments.

In terms of net soil mineral N contribution from biosolids, all profiles except those at Bundaberg showed no evidence of leaching losses below the depths of soil sampling (120–150 cm). Net mineral N in the soil profile generally represented greater
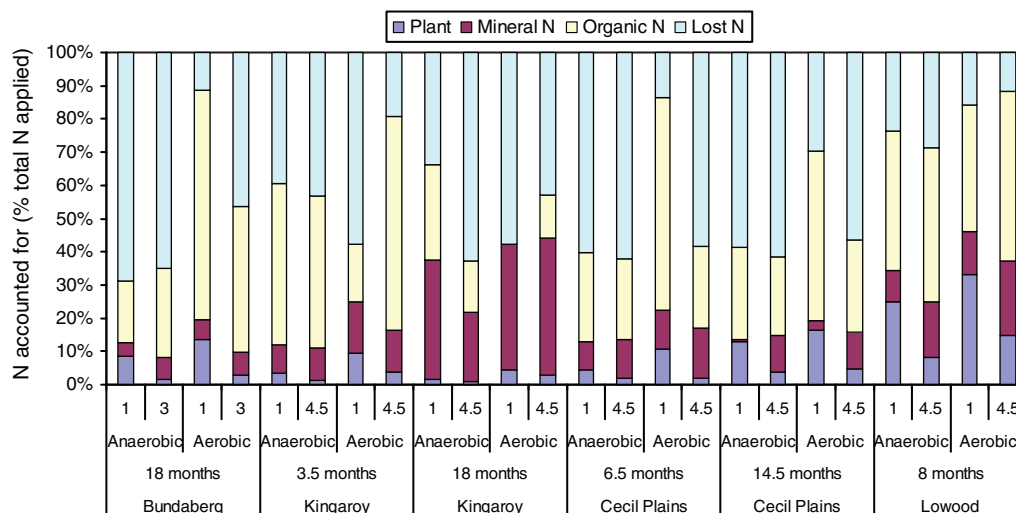
Fig. 4. Partial N budgets for anaerobically and aerobically digested dewatered biosolids applied at five field sites in south-eastern Queensland from 2002 to 2004. Data are expressed as % of the total N applied at two rates of application at each site, with assessment periods ranging from 3.5 months to 18 months after application.

proportions of the added biosolids N than plant N uptake, with the exception of the cut-and-remove forage system at Lowood (where much more N was removed in multiple harvests) and Bundaberg (where deep leaching was clearly evident in the sandy soil; data not shown). In the former, net residual mineral N averaged 95 and 750 kg N/ha for the NLBAR and 4.5NLBAR treatments, i.e. 12% and 20% of the total applied N, respectively. However, at Bundaberg, net residual mineral N averaged only 48 and 200 kg N/ha for the NLBAR and 4.5NLBAR applications, i.e. only 5% and 7% of the total N applied (Fig. 4).

Both the Kingaroy and Cecil Plains sites were sampled after the first and second crop cycles, and while the net N in plant biomass changed little between those sampling events, there was clear evidence of different net mineral N contributions from biosolids in the later samplings at the respective sites. After the first crop (millet at Kingaroy and sorghum at Cecil Plains), net profile mineral N averaged 9% (anaerobic) to 14% of the N applied in the NLBAR treatments, and a similar 11% (anaerobic) to 14% (aerobic) in the 4.5NLBAR treatments (Fig. 4). There was no difference between sites. However, after the second crop (maize at Kingaroy and cotton at Cecil Plains), the net profile mineral N had increased substantially at Kingaroy but either declined (in the NLBAR treatments) or remained static (in the 4.5NLBAR treatments) at Cecil Plains. The totals at Kingaroy ranged from 36% to 38% of the added biosolids N in the NLBAR treatments and from 21% to 41% of the added N in the 4.5NLBAR treatments for the anaerobic and aerobic biosolids, respectively. However, the net mineral N contribution at Cecil Plains was only 1–3% of the added biosolids N in the NLBAR treatments and a consistent 11% in the 4.5NLBAR treatments for the anaerobic and aerobic biosolids, respectively.

Despite the relatively small proportions of applied N recovered in the plant or soil mineral N pools, there was no compelling evidence of the retention of the rest of the applied N in the soil organic N pool (Fig. 4). Variability associated with application of biosolids with commercial spreading equipment (as discussed for mineralisation trial 2), combined with limitations in the sensitivity of soil total N measurement, resulted in greater variability between treatments within sites than in the plant and soil mineral N pools. This variability may have obscured any differences between biosolids types and rates, so to observe general trends, all treatments at a site or sampling event were averaged. This showed there was a relatively consistent 35–45% of the total applied N present in the soil organic N pools, with the exception of the 18-month sampling on the Ferrosol at Kingaroy, where soil organic N represented only 14% of the total applied N.

This combined accounting approach has highlighted some large amounts of lost N at all locations (Fig. 4). These losses were lowest at the cut-and-remove forage site at Lowood (20% of total applied N), but ranged from 40% to 52% of the total applied N for the other sites and sampling occasions. The only site where some losses could be attributed to deep leaching was the freely draining Bundaberg site. Gaseous N losses, via either denitrification or ammonia volatilisation, are expected to be the loss pathway at all other locations (Robinson and Röper 2003; Mendoza *et al.* 2006; Pu *et al.* 2010). Different combinations of soil moisture, pH, and labile C availability will be key factors in determining which N-loss pathway was predominant, with one common factor being a requirement for high concentrations of mineral N (ammonium for volatilisation losses and nitrate for denitrification) in the surface soil. While some reports (e.g. Quemada 1998) suggest ammonia volatilisation losses may be low in field trials when biosolids are incorporated, as in these studies, at least part of the loss for the anaerobic biosolids shown in Fig. 4 is likely to be due to NH_3 volatilisation arising from the very high NH_4-N concentrations in the materials at the time of application (Table 3). If it is assumed that these losses occur during land application before incorporation, as is suggested in the NSW EPA (1997) guidelines, the lost N represents an even higher proportion of the biosolids N that was actually incorporated (i.e. an average of 23% of incorporated N at Lowood and 46–59% at the other sites).

The occurrence of gaseous or leaching losses of biosolids N again suggests that mineralisation of organic N is occurring at rates that are well in advance of plant requirements or the ability of the soil to immobilise that N via microbial activity. This has been demonstrated in both mineralisation trials (Table 6), with rates of organic N mineralisation much greater than estimated in the NLBAR calculation of NSW EPA (1997). Assuming that the lost N (Fig. 4) would have mineralised before denitrification, volatilisation, or leaching, and that plant N was also accumulated from mineral N pools, the amounts of organic N that must have mineralised in the larger scale, biosolids rate field trials were calculated and are presented as proportions of the organic N added in each treatment at each site (Fig. 5). Data again suggest that mineralisation rates of organic N in biosolids are much greater than the 15% (anaerobic) to 25% (aerobic) of organic N that is assumed in the NSW EPA (1997) guidelines to mineralise in the initial crop after application. While statistically significant differences between treatments in rates of organic N mineralisation were detected at Bundaberg, Kingaroy (at 3.5 months), and Cecil Plains (at 6.5 months), all were associated with unusually high or low returns for the lowest rates of application of organic N (the NLBAR rates of aerobic biosolids). Given the variability associated with mechanical spreading at low rates and the subsequent sampling issues, it is perhaps more useful to look at overall trends rather than specific treatments. Average fractions of organic N mineralised ranged from 40% at Kingaroy 3.5 months after application, to 60% at Cecil Plains 6.5 months after application, 50% after 8 months at Lowood, and 56–82% at Bundaberg, Kingaroy, and Cecil Plains after 14.5–18 months.

Summary and conclusions

The high N-mineralisation rates and large recorded losses of applied N highlight some of the knowledge gaps associated with developing sustainable management practices for land application of biosolids. All revolve around the calculation of biosolids application rates based on NLBAR calculation, with three key issues that requiring either further research or a modified set of guidelines compared with those in NSW EPA (1997). The simplest relates to the lack of accounting for existing soil mineral N at the time of biosolids application in determining crop N requirement. Appropriate N-fertiliser application rates are determined by setting a crop yield target, and hence crop N demand, measuring the mineral N stored within the crop root-zone either at or before planting, making an allowance for in-crop mineralisation from soil organic matter, and then calculating the N deficit to be met from fertiliser application. Although the current NLBAR calculation uses a target crop N demand, it does not take into account the background soil mineral N status or mineralisation potential, resulting in greater N application rates. In the case of the maize crops used in the two mineralisation trials at Kingaroy, a target crop-N accumulation of 180 kg N/ha was used in the calculation. This target was achieved in the NLBAR biosolids treatments in trial 1 (i.e. 187 and 186 kg N/ha in the anaerobic and aerobic biosolids treatments, respectively; Fig. 2), whereas accumulation with the lower application rates in trial 2 was 142 and 143 kg N/ha for the same amendments, respectively (Fig. 3). However, in these two trials, the background soil mineral-N status and in-crop mineralisation of soil organic N was able to supply 127 and 85 kg N/ha in the unfertilised control treatments in trials 1 and 2, respectively. Similar effects were recorded in the field studies described in Table 1 and reported in detail in Pu *et al.* (2008). Background soil N reserves were able to supply 37% of the 175 kg N/ha accumulated by sorghum receiving NLBAR rates at Cecil Plains, 42% of the 140 kg N/ha accumulated by millet receiving NLBAR rates at Kingaroy, and 83% of the 190 kg N/ha accumulated in the NLBAR treatments for the sugarcane crop at Bundaberg. Even in the
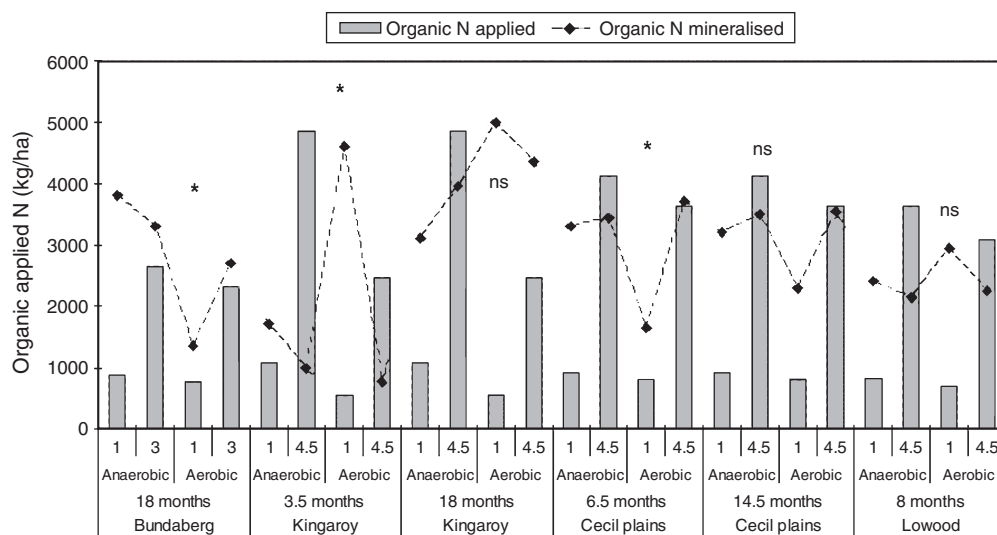
Fig. 5. Rates of applied organic N arising from anaerobically and aerobically digested biosolids applications at rates of 1–4.5 NLBAR at five field sites in south-eastern Queensland, and the calculated percentage of that organic N that had mineralised at assessment times ranging from 3.5 to 18 months after application. Significant differences (*$P < 0.05$) in mineralisation rates between treatments at each site/sampling time are indicated where applicable; n.s., not significant.

cut-and-remove forage system at Lowood, 45% of 500 kg N/ha accumulated by the NLBAR treatments was supplied by background soil N (Pu *et al.* 2008). The result was a generally very low net N contribution from applied biosolids in all studies (Fig. 4), except in the forage system at Lowood. Even if mineralisation rates occurred as predicted, a surplus of mineral N would be expected to accumulate in soil profiles during the crop seasons, which is what was observed in most studies (Figs 2–4).

The second, and perhaps most important issue, is the evidence of extremely rapid rates of mineralisation of organic N in biosolids, relative both to other, more regularly used organic amendments such as manures, composts, and sugarcane mill muds (Tables 4–6; Figs 2 and 3), and also to the estimated 15% (anaerobic) and 25% (aerobic) rates used in the NLBAR calculation (Table 6, Fig. 5). It should be noted that the warmer (and occasionally wetter) climatic conditions in subtropical south-eastern Queensland, compared with those in southern New South Wales where much of the biosolids research in Australia has been conducted (e.g. Salt *et al*. 1996), would be expected to accelerate the rate of organic matter decomposition and therefore N mineralisation. In our studies, the only time we saw significant differences in mineralisation rates between anaerobically and aerobically digested biosolids was in mineralisation trial 1 at Kingaroy, in sampling both at planting (i.e. 8 weeks after application) and at crop harvest some 23 weeks later (Table 6). However, given the uncertainty associated with the fate of N applied as NH_4-N in the anaerobic biosolids and the impact this has on calculations of organic N mineralised, the extent of this difference could not be accurately determined in this study. Collectively, rates seemed to range between 40% and 60% of the applied organic N in the first crop season (Fig. 5), with suggestions that this could have occurred as early as 8 weeks after incorporation of the biosolids in some situations (Table 4). This rapid mineralisation of organic N led to a rapid accumulation of mineral N in the soil profile, as either NH_4-N or NO_3-N, and so the losses from leaching (Bundaberg) or unspecified gaseous N-loss pathways (Table 6, Fig. 4) were not surprising. The low C : N ratio in soil at some sites (9 : 1 for Kingaroy site and 11 : 1 for Lowood site) would have favoured the mineralisation process and reduced any chance of net immobilisation of mineral N once biosolids were incorporated into soil (Green and Blackmer 1995). However, mineralisation rates and subsequent N losses seemed just as high at the Cecil Plains and Bundaberg sites, where the background soil C : N would suggest an increased likelihood of immobilisation. In addition to representing an inefficient use of N in biosolids, the environmental impact of denitrification losses (Pu *et al.* 2010) on greenhouse gas accounts, or the acidifying effects of leaching of NO_3-N deeper into the soil profile (Barry *et al.* 2006) cannot be sustained.

The comparison of partial N budgets from mineralisation trials 1 and 2 (Table 6) suggests that reducing biosolids application rates may be one avenue for improving N-use efficiency in biosolids application programs. However, this represents logistical challenges for land application programs, given the continuous generation and delivery to farms and the requirement to incorporate biosolids with tillage (Barry *et al.* 2006). The latter represents a significant constraint for rainfed grain production systems in southern inland Queensland, where direct-drill tillage systems and extensive reliance on soil moisture stored during crop fallows are significant components of the cropping system (Bell *et al.* 2007). Other options may include the coincidence of biosolids applications with the presence of heavy stubble loads with low N concentrations (i.e. winter cereal stubble) that could potentially facilitate immobilisation of biosolids N in the soil organic matter pool for later release and utilisation. These management options require research validation.

Finally, the contrast between N-mineralisation rates in biosolids and other organic amendments was marked (Tables 4–6; Figs 2 and 3), and again highlights the need to better understand factors that characterise mineralisation of organic materials in broadacre agriculture. Simple C : N ratios have often been used as a guide, but while this indicates greater rates of N mineralisation in biosolids than in other amendments (Table 3), the relative mineralisation rates among other amendments are not well correlated with the C : N ratios. For example, the slower mineralisation rate of the composted feedlot manure and the apparently more rapid mineralisation rates of the mill mud (Table 5), compared with the other non-biosolids amendments in the respective studies, suggest that a better understanding of N mineralisation in these materials is also an area for future research.

Acknowledgments

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