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FINAL REPORT – SRDC PROJECT BSS199
IMPROVING THE MANAGEMENT OF ACID AND SODIC SOILS WITH
GREEN TRASH RETENTION USING CALCIUM-BASED
AMELIORANTS/PRODUCTS

by

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SUMMARY

Although the advantages of leaving cane residues on the harvested land have been well documented, and nutrient cycling in relation to green-cane trash blanketing has been well examined within a Cooperative Research Centre for Sustainable Sugar Production (CRC Sugar) program activity, little work has been done to investigate the effects of trash management on soil chemical properties and the ameliorative advantages of applying Ca-containing amendments to crop residues. As lime, lime/gypsum applications are generally used to ameliorate acid soil conditions, a Ca-source followed by leaching is the principal strategy for managing sodic soils, and mill by-products (mill mud and mill ash) are often applied to plant cane, it was considered pertinent to investigate the effects of these and other amendments on the decomposition of trash on sugar industry soils.

This project aimed at quantifying the effects of trash retention on soils acidity, examining the efficacy of Ca-containing amendments and trash for ameliorating sodic soils, assessing the effect of Ca-based amendments on trash decomposition and nutrient availability, and quantifying trash decomposition and C and N relationships. To do this, the project comprised four distinct but interlinked facets and included a series of laboratory investigations, glasshouse experiments and a field trial.

Green-cane trash retention has an effect on soil properties. Increases in organic carbon due to trash retention result in improved charge characteristics of the soil, but with acidic cations (H^+ and Al^{3+}) predominantly retained on the exchange complex. Lime application would, however, neutralise this effect and result in an increased capacity of the soil to retain other nutrients, such as K^+ , Ca^{2+} and Mg^{2+} . As the ash alkalinity of sugarcane is small compared to that of other litter materials, trash does not contribute significantly to neutralising acidity generated by nitrogen mineralisation.

Decomposition of sugarcane is generally unaffected by soil sodicity. Cane trash and ameliorants such as molasses and soil organic matter are all important in the management of sodic soils. All forms of organic additions examined improved soil structural properties and reduced sodicity or the tendency of soil to become or remain sodic. There were no major interactions between organic matter and Ca-containing amendments on ameliorating sodic conditions; they all had positive effects irrespective of each other.

Calcium-based amendments such as lime and mill mud have the potential to accelerate the decomposition of sugarcane trash when applied to the surface of the trash layer. When comparing the effectiveness of the amendments in assisting decomposition, millmud repeatedly outperformed the other amendments. Other amendments, such as $Ca(OH)_2$ and KOH , did have the ability to enhance the decomposition process to some extent, but they are not considered suitable for widespread use in the industry due to their corrosive nature. Importantly, the enhanced trash decomposition that occurred due to lime millmud, mill ash and lime applied to trash in pots under semi-controlled conditions was also found to be applicable in the field. The differences in the rates of decomposition in the simulated subtropical and moist tropical conditions suggest that environmental conditions involving temperature and the amount of water (rain and/or irrigation) are important factors within this trash/ameliorant system.

Lime applied to the surface of the trash layer did have the ability to ameliorate acidic conditions in the underlying soil to some extent. This has important implications for certain soils when minimum tillage is practiced.

Although millmud and mill ash applications to trash resulted in significantly higher amounts of N, P and K removed by the harvested biomass, these elevated values could either have been the result of the nutrients within the amendments themselves, or due to enhanced decomposition of the trash, or a combination of both of these effects. The effect on nutrient uptake by sugarcane when lime was applied to trash was less apparent. Environmental effects (water, temperature and soil properties) are again considered important factors in relation to the overall nutrient availability in these amended trash systems

The small-scale decomposition experiments showed that decomposition of trash used in such trials is unaffected by factors such as drying, rough grinding or whether the trash is green or senesced. However, as decomposition is affected by soil moisture and placement, such factors need to be carefully controlled. Importantly, measurement of decomposition as CO₂ loss appears to be a more reliable and sensitive method than measuring dry matter remaining in such small-scale investigations. Although millmud can contribute relatively large amounts of N, applications in the presence of trash may reduce N availability initially due to microbial immobilisation.

Generally, the amendments, such as lime, millmud and millmud/mill ash mixtures applied to the surface of a trash layer have positive results for sugarcane production. Lime applied to trash has the ability to counter acidity generated by the mineralisation of N from the organic matter. Sources of organic matter (trash, millmud, molasses, etc), whether present individually or in combination with each other, have the ability to ameliorate sodic soil conditions. Ameliorants such as lime and millmud have the potential to accelerate trash decomposition when applied to the surface of the trash layer. Ultimately more efficient return of organic matter to the soil will improve amelioration of soil acidity and sodicity, and result in improved nutrient- use efficiency.

Although some technology transfer has occurred in terms of papers, presentations and posters, further extension of the concept is proposed. In particular, a paper entitled 'Cane trash and Ca-based soil amendments: A combination too good to miss' will be presented at the 2004 ASSCT conference. Further extension will make use of grower meetings and workshops.

1.0 BACKGROUND

With the move away from burning sugarcane to the retention of crop residue in much of the Australian sugar industry, major implications for water, carbon and nutrient dynamics have been identified and investigated in various research programs. However, little attention has been directed towards investigating the short and longer-term impacts of trash layers on changes in chemical properties. In particular, there have been reports in literature of both increases and decreases in soil pH with the addition of organic matter to agricultural soils. In this light, local evidence suggested that decreases in soil pH have occurred in at least some of the long-term trash-retention trials conducted at various sites in the industry. Such increases in soil acidity could have important implications for both the acid and sodic sugarcane-producing soils. A decrease in soil pH associated with trash could have negative implications for already acid soils, but could be advantageous to alkaline sodic soils. In both cases it would be pertinent to consider the use of Ca-containing amendments (lime and/or gypsum) in association with trash for ameliorative purposes.

In acid soils, the surface applications of lime (or lime/gypsum) to trash would be considered important to counteract any increases in soil acidity. However, lime could also increase the decomposition rate of trash and possibly contribute to improved nutrient use efficiency in such systems. In sodic soils, the organic acids and carbon dioxide produced during decomposition of trash may improve the efficacy of various Ca-containing amendments for improving soil physical conditions.

2.0 OBJECTIVES

This project aimed to develop a set of recommendations for cane growers to ensure full utilisation of the interactive processes between trash and Ca-containing amendments.

The specific objectives of the project were to:

1. Quantify the effects of trash retention on soil acidity and sodicity;
2. Determine the influence of Ca-containing amendments on trash decomposition and nutrient availability under various environmental conditions;
3. Determine the influence of trash decomposition on the efficacy of Ca-containing amendments in improving acid and sodic soils;
4. Assess strategies for maximising the beneficial interactions between trash and Ca-containing amendments.

All the objectives of the project have been achieved.

The underlying theme of the project was to ensure the optimum utilisation of the interaction between trash and Ca-containing amendments to enable sustainability, which encompasses both productivity and environmental issues. This implied that strategies would be sought to counteract any negative effects on soils properties due to green-cane trash blanketing (GCTB) and to translate any positive interactive effects into increased productivity.

The methodology used in this project consisted of four separate, yet interlinked facets (Figure 1) that aimed to answer the following questions:

- Does the presence of trash affect soil pH?
- If so, why?
- What process is involved?
- What liming strategies are applicable for countering any negative effects?
- What strategies could be used to harness any positive effects?
- Would these strategies have an effect on nutrient availability?

Objective 1 – Quantify the effects of trash retention on soil acidity.

Facet (i) of the project was specifically aimed at determining the effect of trash retention on the chemical properties of the underlying soil. This section of the project was achieved by quantifying the net proton input associated with long-term burn/trash trials, determining changes in soil properties of the soils associated with the two systems, and assessing the effect on soil of the water-soluble organic and inorganic components derived from plant litter.

Objective 2 – Determining the influence of Ca-containing amendments on trash decomposition and nutrient availability in improving acid and sodic soils.

Facets (iii) and (iv) of this project were aimed at determining the influence of Ca-containing amendments on trash decomposition. A range of amendments was assessed and included lime, gypsum and the important sugar industry by-products – millmud and mill ash. Other possible amendments were also included in the studies. The assessments involved pot experiments and laboratory investigations and a field trial. Importantly, lime and millmud applied to the surface of the trash layer were found to be useful in ameliorating the underlying soil and capable of improving the nutrition of sugarcane grown within such trash amended systems.

Objective 3 – Determining the influence of trash on the efficacy of Ca-containing amendments in improving acid and sodic soils.

Long-term trash/burn trials provided the opportunity to study the effects of trash retention on soil chemical properties. Increases in organic C with trash retention and corresponding increase in charge characteristics inherently improves nutrient retention, especially in low-CEC tropical soils. Laboratory based experiments were important in assessing the quality of leachates originating from litter materials. Facets (ii), (iii) and (iv) dealt with various aspects of the interaction between trash and Ca-containing amendments. It appears that millmud applied to the surface of a trash blanket has a number of positive results (accelerated decomposition, ameliorative effects and possible improvements in nutrient-use efficiency).

Objective 4 – Assessing strategies for maximising the beneficial interactions between trash and Ca-containing amendments.

Facet (iii) in particular investigated practical means of maximizing the benefits of applying Ca-containing amendments to trash. Amendments such as lime, millmud and mill ash (routinely available in the sugar industry) were assessed. Each of these in

combination with trash was found to have beneficial effects. These amendments applied to the surfaces of trash blankets have shown to have ameliorative, nutritional and environmental benefits.

3.0 PARTIES INVOLVED AND LINKAGES

This project was established as a joint initiative involving scientists from BSES, CSIRO Land and Water, and CSR Sugar Ltd. The common association of the participating scientists was the CRC for Sustainable Sugar Production (CRC Sugar). Importantly, the scientists that were involved had experience in aspects of trash management, trash decomposition, soil science, soil fertility and crop nutrition.

The project had important linkages to several other projects, funded either by CRC Sugar or SRDC. These included:

- CRC Sugar funded: 2.3.8 – Opportunities for better nutrient supply and uptake;
- CRC Sugar funded: 2.2.2.1 – Nutrient cycling in relation to trash-management systems;
- CRC Sugar funded: 2.3.3 – Different aspects of soil acidity;
- CRC Sugar funded: 2.3.2 – A tool kit for improved diagnosis and management of sodic soils;
- SRDC funded: BSS197 – Products and mechanisms for amelioration of sodic soils;
- SRDC funded: CSR24 – Improving the environment for sugarcane growth through the amelioration of soil acidity.

4.0 METHODOLOGY

As indicated in the schematic representation of the project (Figure 1) the project was divided into four facets within two main sections. The first section related to the effect of trash retention (Facet i.). The second section related to aspects of Ca-containing amendments applied to trash (Facets ii, iii and iv).

The four facets aimed at:

- i. Quantifying the effects of trash retention on soil acidity;
- ii. Examining the efficacy of Ca-containing amendments and trash for ameliorating sodic soils;
- iii. Assessing the effect of Ca-based amendments on trash decomposition;
- iv. Quantifying trash decomposition and C and N relationships.

The responsibility for the various facets lay with different scientists working within the project. The names of the appropriate persons are shown in Figure 1. The methodology associated with each of these facets (i, ii, iii and iv) is detailed within relevant sections (Sections 5, 6, 7 and 8, respectively).

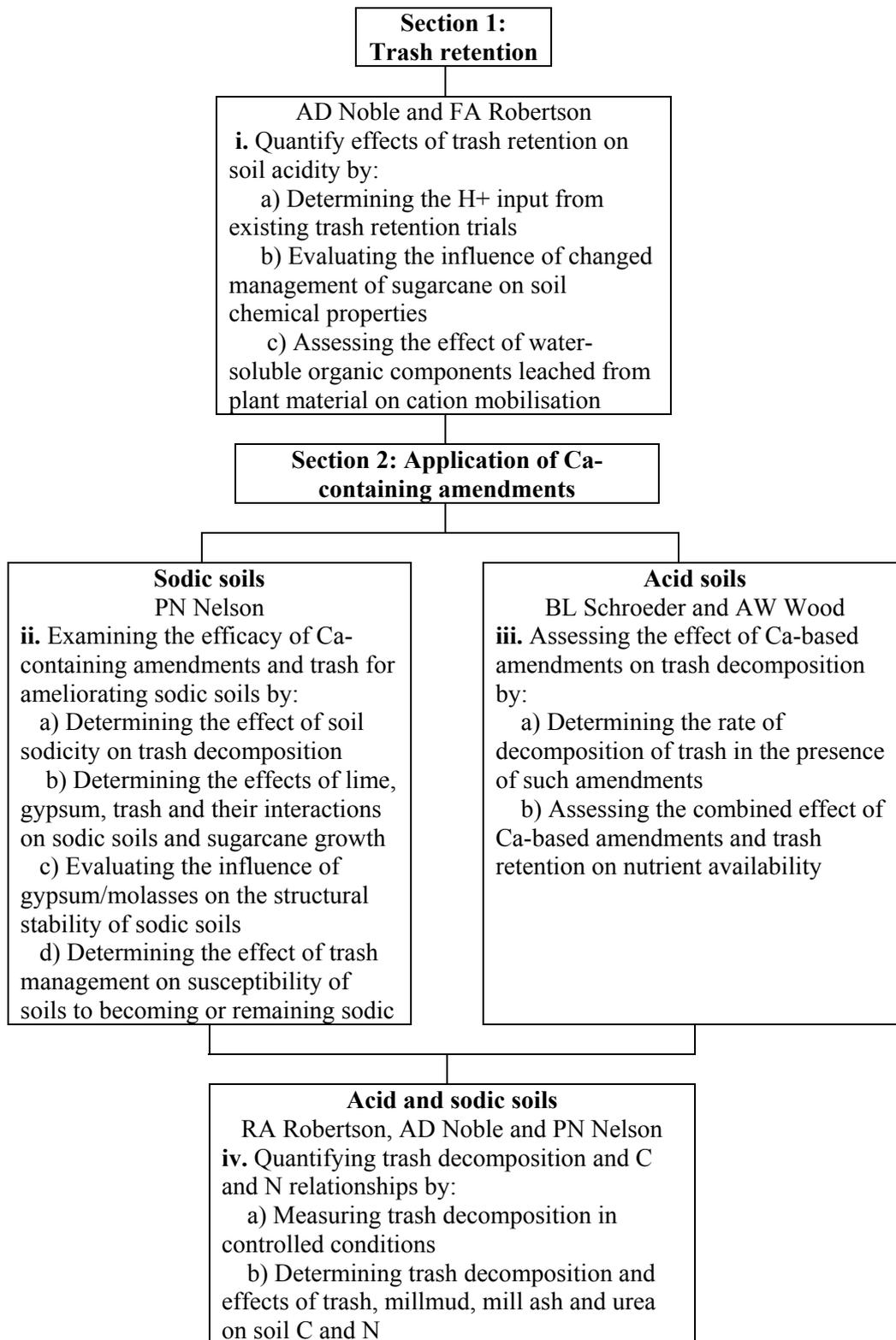


Figure 1 Schematic representation of the various facets of the project

5.0 EFFECTS OF TRASH RETENTION ON SOIL ACIDITY

5.1 Introduction

Under intensive sugarcane production systems, continual acidification of the soil resource base is likely to occur through the use of ammoniacal nitrogen fertilisers inputs, base removal as a result of crop export, and the general high rainfall environment under which sugarcane is grown that facilitates the losses of basic cations through leaching (Moody and Aitken 1995). Whilst Australian sugarcane cultivars are relatively tolerant of acidity (i.e. low pH and Al phytotoxicity), they are highly intolerant of low levels of soluble Ca^{2+} (Hurney 1971; Ridge *et al.* 1980; Haysom *et al.* 1986; Hetherington 1986). Reghenzani (1993) in a survey of soil nutrient levels estimated that 67% of the soils in north Queensland had suboptimal levels of exchangeable Ca^{2+} , and the remediation of a Ca^{2+} deficiency can result in significant increases in yield (Hurney 1971; Ridge *et al.* 1980; Haysom *et al.* 1986; Quinan and Wood 1989, 1993; Noble *et al.* 2000). It has been clearly shown that long-term sugarcane monoculture results in a gradual decline in soil fertility that is associated with soil acidification (Bramley *et al.* 1996; Garside *et al.* 1997; Moody and Aitken 1997).

Whilst the alleviation of acid soil infertility of surface horizons by regular additions of lime effectively neutralises a high proportion of acidity generated through anthropogenic activities, the remediation of subsoil acidity is more problematic. Considerable work has been conducted in which lime has been mechanically mixed into the soil to depth (0.5-1 m) in order to neutralise acidity (McKenzie and Nyborg 1984; Hammel *et al.* 1985). Such treatments have included: chisel ploughing in combination with rotary tillage (Doss *et al.* 1979); subsoil lime injectors both mechanical (Anderson and Hendrick 1983) and pneumatic (Saayman and van Huyssteen 1981); mouldboard ploughing (Lutz and Jones 1971); and various mechanical procedures in combination with lime and gypsum (Farina and Channon 1988). These methods have all shown that the neutralisation of subsoil acidity has been effective in establishing a more hospitable rooting environment, which has usually translated into increased yields particularly in years of below average rainfall. The economic efficacy of such treatments is determined by commodity price structures, soil type and tillage costs and may not be a feasible option for all situations (Sumner 1995). However, in view of the residual value of lime, the economics of lime application, whether for top or subsoil amelioration, should be considered over more than one crop year (Mallawaarachchi *et al.* 1998).

The adverse effects of long-term sugarcane monocultures have resulted in a decline in the productive capacity of soils and, hence, productivity (Garside *et al.* 1999). In 1993, a series of rotation break trials was established on long-term sugarcane monoculture fields to determine if rotations caused changes in the chemical, physical and biological properties of the soil that would reverse the negative effects of declining yield (Garside *et al.* 1997). Significant increases in cane and sugar yield have been observed following the implementation of a break crop management strategy (Garside *et al.* 1999).

With respect to aspects of changes in soil fertility associated with GCTB or the influence of crop rotations under Australian production systems, the focus has been on rates of decomposition, the carbon and nitrogen cycles, and changes in the chemical status of soils (Spain and Hodgen 1994; Robertson and Thorburn 2000). Of importance in assessing the

impact of changed management practices on the soil resource base is quantification of possible changes in exchange properties of soils associated with different management regimes. As part of this study, changes in chemical attributes and surface charge characteristics of soils associated with two forms of changed management in a sugar production system were quantified.

Three interlinked but separately initiated studies were undertaken to:

1. Quantify the net proton input associated with long-term trash retention systems through the evaluation of changes in soil-chemical attributes from long-term burn/trash retention trials from a range of sites within the eastern sugar-growing belt.
2. One of the dominant impacts of trash retention on soil chemical properties is the enhanced organic matter content of soils. Changes in soil-chemical properties were quantified using two contrasting production systems that are currently being promoted in the Australian sugar industry; long-term trash retention versus burnt system, and a grass-ley production system versus continuous monoculture of cane. In addition to the standard chemical attributes measured, surface charge was characterised, as this is one of the dominant soil attributes affected by organic matter conservation.
3. Assess the effect of water-soluble organic and inorganic components emanating from different surface-applied plant materials (intact or ashed) on the mobility of cations and the remediation of acidity. In addition, the chemical composition of the water-soluble leachate prior to and after leaching through a soil column was compared.

5.2 Net H⁺ input from existing trash retention trials

5.2.1 Methods and materials

Site characteristics and treatment

Burnt and GCTB systems were compared in three field sites at three locations along the eastern sugar belt; these have contrasting climatic conditions (Table 1). The Tully and Mackay sites were established in 1990 and 1992, respectively, whilst the Harwood site was established in 1995. Robertson and Thorburn (2000) give further details of their management. All three trials were planted to cultivar Q124.

Table 1 Selected site characteristics of trials used in the assessment of net proton input under trash retention and burn management systems

	Tully	Mackay	Harwood
Location	17.56 S; 145.56 E	21.10 S; 149.07 E	29.50 S; 153.20 E
Year established	1990	1992	1995
Date sampled	November 1997	August 1998	January 1998
Soil texture	Clay	Loam	Clay/loam
Mean annual rainfall (mm/yr)	4067	1668	1021
Previous cropping history	Sugarcane	Sugarcane	Vegetables, sugarcane

Soil samples

Soil samples were collected by taking 12 cores (3.0-3.4 cm internal diameter) in two regular transects from the top of each row and across the inter-row, with each core being sectioned into the depth layers 0-2, 2-5, 5-10 and 10-25 cm. Soils were air dried and passed through a 2-mm sieve.

Ash alkalinity determinations

Composite sugarcane trash samples were collected from each of the three trial sites and submitted for chemical analysis and ash alkalinity. Samples were dried to a constant weight at 60°C in a forced-draft oven before being ground to pass through a 2-mm mesh and then Tema milled to a powder. The elemental composition of the leaf material was determined by X-ray fluorescence spectrometry (XRF) on four replicated pressed pellets of the milled trash (Norrish and Hutton 1977). The ash alkalinity was determined by the difference between total cations and anions: $\Sigma (\text{Ca}^{2+} + \text{Mg}^{2+} + \text{K}^{+} + \text{Na}^{+}) - \Sigma (\text{SO}_4^{2-} + \text{H}_2\text{PO}_4^{-} + \text{Cl}^{-})$.

pH buffer capacity determination, organic carbon and particle size analysis

We used a modified discontinuous batch method of James and Riha (1986) to determine the acid and base neutralizing capacity (ANC and BNC, respectively) of soils collected from each of the trial sites and depth intervals. The determination of pH buffer capacity was undertaken on all samples collected. In brief, 2 g of air-dried soil that had been sieved to pass through a 2-mm mesh were shaken in polypropylene tubes containing a total liquid volume of 25 mL and included 0, 0.5, 2.0, 4.0, 10.0 or 20.0 mL of 0.001M HNO₃ or 0.001M KOH, 1.0 mL KNO₃ and 250 µL chloroform. After equilibrating on an end-over-end shaker for 22 h at 25°C, the suspensions were centrifuged and the supernatant pH measured. pH buffer curves were constructed for both the ANC and BNC by taking the inverse of the linear section of plots of acid/base addition versus pH.

Organic carbon (OC) content and particle size distribution were determined on composite samples from each depth interval for each treatment. Organic carbon was determined by wet oxidation using the Walkley and Black (1934) method as described in Method 6A1 in Rayment and Higginson (1992). Particle size was determined using the methodology of Coventry and Fett (1979).

5.2.2 Results and discussion*Ash alkalinity determination*

The simplest method of determining the ash alkalinity in plant material is by titration of an acidified solution containing the ashed sample with standardised base (Jarvis and Robson 1983). Whilst this method is sufficient in most cases to give an accurate assessment of this parameter, it has the inherent potential error associated with the volatilisation of S and Cl during the ashing process (Jungk 1968; Kennedy 1992). Under such circumstances, losses of these anions would result in an over-estimation of the ash alkalinity. A far more accurate assessment of ash alkalinity can be undertaken by subtracting the total cations from the total anion complement of the plant material as determined by XRF. The percent concentrations of both cations and anions are converted to their respective equivalent concentrations of the species presented in equation 1.

$$\text{Ash alkalinity (cmol/kg)} = \Sigma(\text{Ca}^{2+} + \text{Mg}^{2+} + \text{K}^{+} + \text{Na}^{+}) - \Sigma(\text{SO}_4^{2-} + \text{H}_2\text{PO}_4^{-} + \text{Cl}^{-}) \quad (1)$$

The elemental composition and the ash alkalinity of the three trash samples of Q124 collected from the three trial sites are presented in Table 2. The Ca concentration from the trash samples collected from the Mackay site were higher than the other two sites, whilst the K concentration was higher in the Tully samples. The Si concentration in the Tully sample was considerably lower than those sampled at the Mackay and Harwood sites, possibly reflecting the higher leaching environment under which cane is produced (Table 2). There was a highly significant simple linear relationship between the Ca concentration in the stalk and ash alkalinity over all sites (Figure 2).

Table 2 Elemental composition and ash alkalinities of cane trash samples collected from each of three trial sites. Each value is the mean of four replicates

Site	Ca	Mg	K	Na	Al	Si	P	Cl	SO ₄	S	Total S
	(mg/g)										
Mackay	3.06	1.31	3.49	0.48	1.99	25.9	0.44	1.31	0.33	0.34	0.64
Harwood	1.95	1.34	2.50	0.01	2.01	26.3	0.34	0.81	0.28	0.34	0.59
Tully	1.56	1.08	6.28	0.01	3.17	14.5	0.49	2.66	0.35	0.28	0.60

Site	Zn	Cu	Fe	Mn	Ash alkalinity
	ppm				(cmol _c /kg)
Mackay	32	24	2124	236	28.12
Harwood	27	14	1760	183	20.80
Tully	24	15	1174	183	20.46

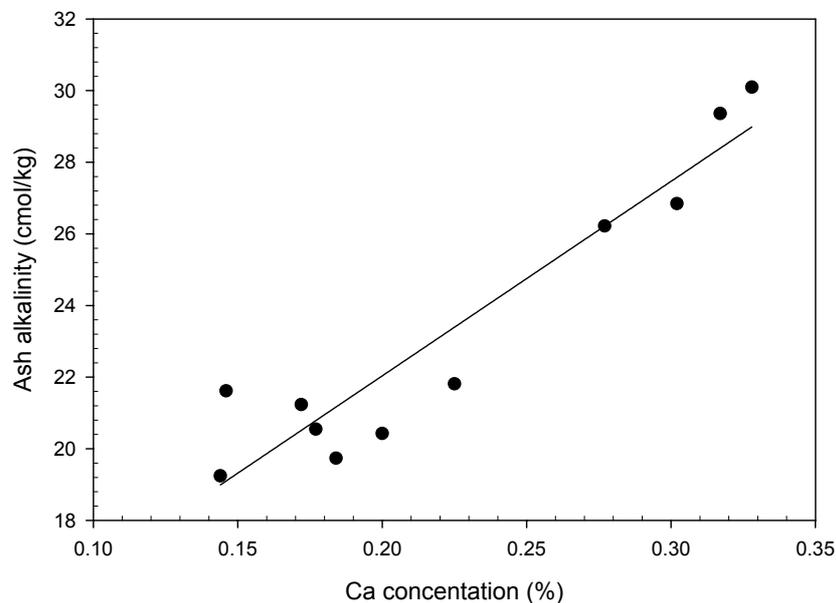


Figure 2 Relationship between Ca concentration in the stalk and ash alkalinity of samples collected from Harwood, Mackay and Tully. Equation: Ash alkalinity (cmol/kg) = 11.16 (\pm 1.46) + 54.32 (\pm 6.24)Ca conc. (%): $r^2=0.894$.

Ash alkalinity has been used extensively to describe the acid-base balance in foods, grains and fodder crops and is a measure of the organic-anions that are produced internally by the plant in order to balance excess cation uptake. It is the decomposition of these organic anions that influence the proton pool. The ash alkalinity of the trash material can be expressed as CaCO₃ equivalents if it is assumed that 1 mol H⁺ requires 50 g CaCO₃ to be neutralised. Consequently, the net amount of alkalinity in the form of CaCO₃ equivalents associated with the trash ranges from 10.23 to 14.6 kg CaCO₃/tonne dry matter. These rates of alkalinity addition on an annual basis through trash retention, are small compared to the litters collected from some plant species (i.e. *Melia azedarach* 126 kg CaCO₃/tonne dry matter) (Noble *et al.* 1996) and, therefore, do not make a significant contribution to the net acid input on an annual basis. Robertson (1998) reported net annual trash additions to the Harwood, Mackay and Tully sites of 8.76, 11.50 and 6.93 t/ha, respectively. Based on a mean CaCO₃ equivalent of 12.41 kg/t of dry matter, the total potential alkalinity release from trash applied would be equivalent to 108, 142 and 86 kg/ha/yr for the three sites.

Table 3 pH buffer capacity values for soils collected from the three trash management trials.

Location	Treatment	Depth (cm)	Acid pHBC	Base pHBC	Mean pHBC
			(mmol H ⁺ OH ⁻ /kg.unit pH)		
Mackay	GCTB	0-2	25.4	32.8	29.1
Mackay	GCTB	2-5	22.8	28.0	25.4
Mackay	GCTB	5-10	26.3	26.5	26.3
Mackay	GCTB	10-25	28.2	26.3	27.2
Mackay	Burnt	0-2	22.0	25.6	23.8
Mackay	Burnt	2-5	22.8	24.2	23.5
Mackay	Burnt	5-10	24.5	24.2	24.3
Mackay	Burnt	10-25	27.2	24.6	25.9
Harwood	GCTB	0-2	45.6	55.8	50.7
Harwood	GCTB	2-5	49.6	50.0	49.8
Harwood	GCTB	5-10	51.0	49.7	50.3
Harwood	GCTB	10-25	49.7	49.2	49.4
Harwood	Burnt	0-2	44.8	50.2	47.5
Harwood	Burnt	2-5	52.1	48.7	50.4
Harwood	Burnt	5-10	50.2	48.7	49.4
Harwood	Burnt	10-25	52.1	49.0	50.5
Tully	GCTB	0-2	34.6	28.5	31.5
Tully	GCTB	2-5	40.2	22.3	31.2
Tully	GCTB	5-10	41.5	20.2	30.8
Tully	GCTB	10-25	41.7	19.6	30.6
Tully	Burnt	0-2	35.5	19.9	27.7
Tully	Burnt	2-5	38.3	19.6	28.9
Tully	Burnt	5-10	37.9	20.6	29.2
Tully	Burnt	10-25	40.8	19.5	30.1

pH buffer capacity determination

Soil pH buffer capacity (pHBC) values for each of the depth intervals collected at the three sites are presented in Table 3. pHBC is calculated from the inverse of the slope of the linear region of curve of the plot between pH (y) and acid/base added (x). This linearity occurs between pH 4 and 6. An example of the ANC and BNC curves for the 0-2 cm depth interval of the Mackay soil is presented in Figure 3.

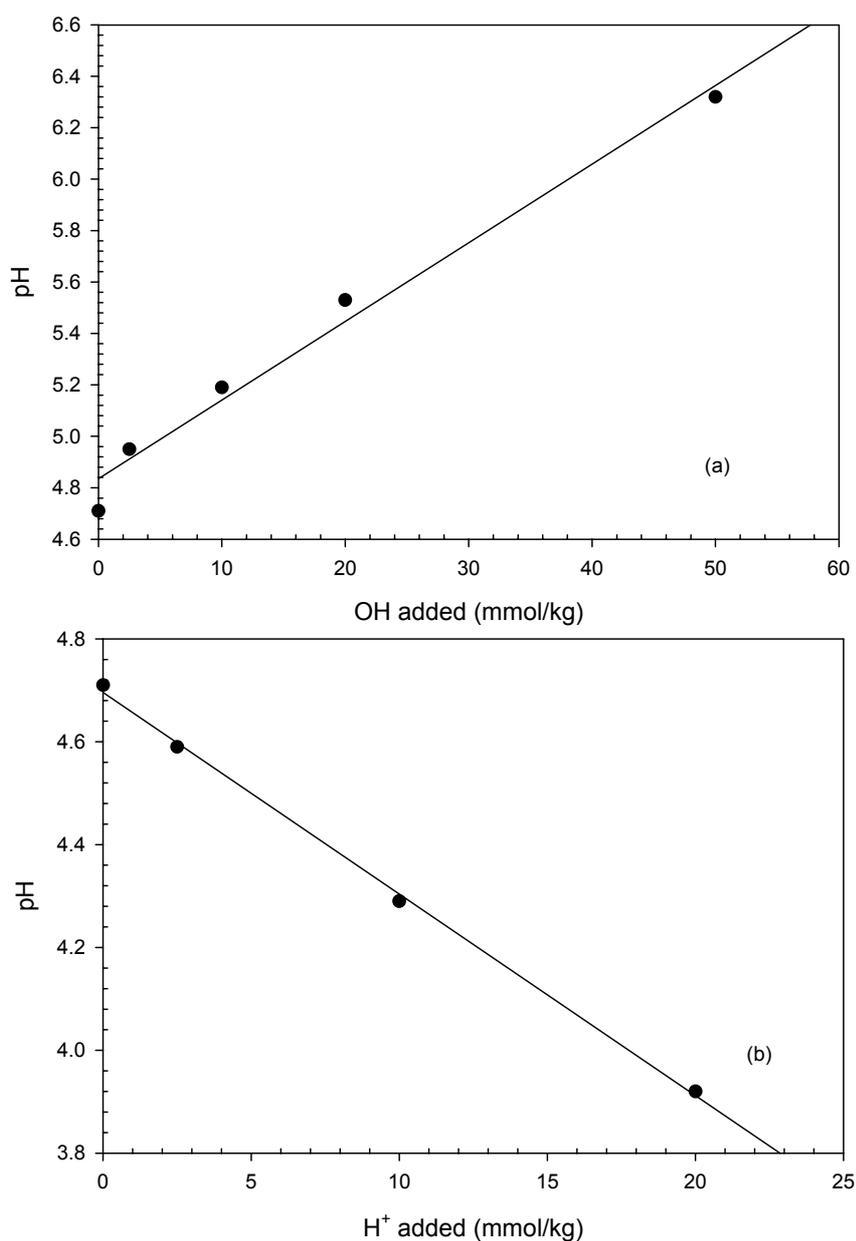


Figure 3 pH buffer curves for (a) base and (b) acid additions for the Mackay 0-2 cm under a green cane trash blanket treatment

pHBC values in the surface 0-2 cm were generally slightly higher under the GCTB treatments when compared to the burnt. These differences diminished with depth. The

pHBC values as determined by either acid or base addition were similar in the case of the Mackay and Harwood sites, but noticeably different in the case of the Tully site (Table 3). This could not be explained on the basis of clay content, as both the Tully and Harwood samples had similar clay contents (Table 4). Whilst no mineralogical characterisation was undertaken of samples, it is plausible that this disparity may in part be associated with the presence of sesquioxidic surfaces that are common to highly weathered tropical soils. As the pH of these soils declines, ANC increases through the consumption of protons with a concomitant increase in positive surface charge. This would result in a higher capacity to buffer shifts in pH associated with acid addition.

Table 4 Soil organic carbon (OC) and particle size distribution of soils collected for pH buffer capacity determination from Mackay, Harwood and Tully

Location	Treatment	Depth (cm)	Mean pHBC	OC	Clay	Silt	C Sand	F Sand
			(mmol/kg.unit pH)					
Mackay	GCTB	0-2	29.1	2.5	15.9	27.0	12.6	44.4
Mackay	GCTB	2-5	25.4	1.5	17.7	26.7	10.9	44.6
Mackay	GCTB	5-10	26.3	1.3	18.5	24.6	11.1	45.8
Mackay	GCTB	10-25	27.2	1.1	18.1	24.8	11.6	45.5
Mackay	Burnt	0-2	23.8	1.5	15.6	27.2	13.8	43.5
Mackay	Burnt	2-5	23.5	1.3	17.4	25.4	13.1	44.1
Mackay	Burnt	5-10	24.3	1.2	17.9	25.4	12.9	43.8
Mackay	Burnt	10-25	25.9	1.2	18.1	26.3	11.5	44.1
Harwood	GCTB	0-2	50.7	3.2	35.6	46.7	4.4	13.3
Harwood	GCTB	2-5	49.8	2.7	37.6	46.9	3.7	11.8
Harwood	GCTB	5-10	50.3	2.6	37.1	47.3	3.7	11.9
Harwood	GCTB	10-25	49.4	2.4	38.1	46.9	3.5	11.5
Harwood	Burnt	0-2	47.5	3.0	34.3	44.5	4.8	16.4
Harwood	Burnt	2-5	50.4	2.6	35.0	45.3	5.4	14.4
Harwood	Burnt	5-10	49.4	2.5	37.0	43.1	6.1	13.8
Harwood	Burnt	10-25	50.5	2.7	36.9	44.1	5.7	13.4
Tully	GCTB	0-2	31.5	3.1	33.0	38.8	7.0	21.2
Tully	GCTB	2-5	31.2	1.7	37.3	36.8	6.2	19.7
Tully	GCTB	5-10	30.8	1.4	40.7	32.0	6.3	21.0
Tully	GCTB	10-25	30.6	1.3	39.3	35.9	5.6	19.4
Tully	Burnt	0-2	27.7	1.6	33.3	32.9	6.1	27.6
Tully	Burnt	2-5	28.9	1.4	36.6	33.2	5.4	24.7
Tully	Burnt	5-10	29.2	1.3	36.7	32.7	5.4	25.1
Tully	Burnt	10-25	30.1	1.3	37.8	31.9	5.3	25.0

It is generally agreed that pHBC is a function of soil pH and clay and organic matter content. To determine whether these routinely measured soil attributes could explain measured mean, acid and base pHBC, we undertook simple linear regression analysis. Based on the above difference in the buffer capacity associated with the addition acid or base between the Tully soil and both the Harwood and Mackay soils, the latter two soils were grouped as a single data set and Tully data treated separately. Regression and correlation coefficients are presented in Table 5. It is clearly evident that in the combined Harwood/Mackay data set, regardless of whether mean, acid or base pHBC was used, OC

and clay content were highly significantly ($P < 0.001$) correlated (Table 5). Clay content tended to explain a greater degree of the variation than OC. In the case of the Tully data set, acid pHBC was best described by clay content. In contrast, base pHBC was best described by OC (Table 5). This would tend to support the above argument based on differences in mineralogy between the soils from the wet tropics and those from regions outside this area. In the case of the Tully site, the soil appears to be exhibiting ‘variable charge’ characteristics. The term ‘variable charge’ is used to describe organic and inorganic soil constituents with reactive surface groups whose charge varies with pH, ionic concentration and composition of the soil solution. Such groups are the surface carboxyl, phenolic and amino functional groups of organic materials in soils, and surface hydroxyl groups of Fe and Al oxides, allophane and imogolite. The hydroxyl surface groups are also present on edges of some phyllosilicate minerals such as kaolinite, mica, and hydroxyl-interlayered vermiculite. The variable charge is developed on the surface groups as a result of adsorption or desorption of ions that are constituents of the solid phase, i.e. H^+ , and the adsorption or desorption of solid-unlike ions that are not constituents of the solid. This explains the strong relationships between acid pHBC and clay (i.e. sesquioxide surfaces) and the base pHBC and OC, as organic matter has only ability to buffer increases in OH^- .

Table 5 Regression coefficients for selected linear regressions for mean, acid and base pHBC for the Tully (n=8) and combined Harwood/Mackay (n=16) data sets. Values in parenthesis are the standard errors

Site	pHBC	Constant	OC (%)	Clay (%)	Variance accounted for by model (%)
Harwood/ Mackay	Mean	6.31 (4.41)	15.10 (2.00)		78.8 ^{***}
	Mean	4.14 (1.51)		1.25 (0.05)	97.4 ^{***}
	Mean	2.61 (0.97)	3.91 (0.78)	1.00 (0.06)	99.0 ^{***}
	Acid	7.29 (5.75)	14.34 (2.61)		66.0 ^{***}
	Acid	2.60 (1.63)		1.28 (0.06)	97.1 ^{***}
	Acid	2.57 (1.79)	0.07 (1.44)	1.28 (0.11)	96.9 ^{***}
	Base	5.33 (3.25)	15.86 (1.47)		88.4 ^{***}
	Base	5.69 (2.56)		1.21 (0.08)	92.4 ^{***}
	Base	2.65 (0.90)	7.74 (0.73)	0.73 (0.05)	99.2 ^{***}
Tully	Mean	28.49 (1.37)	0.95 (0.79)		5.9 ns
	Mean	23.96 (7.06)		0.16 (0.19)	-
	Mean	6.11 (4.48)	2.48 (0.46)	0.54 (0.11)	81.4 ^{**}
	Acid	43.73 (2.32)	-3.00 (1.34)		36.7 ns
	Acid	2.79 (4.52)		0.98 (0.12)	90.0 ^{***}
	Acid	5.59 (7.24)	-0.39 (0.75)	0.92 (0.17)	88.6 ^{**}
	Base	13.26 (0.78)	4.89 (0.45)		94.3 ^{***}
	Base	45.10 (14.40)		-0.65 (0.39)	19.9 ns
	Base	6.63 (5.56)	5.35 (0.57)	0.16 (0.13)	94.7 ^{**}

** significant at the $p < 0.01$ level; *** significant at the $p < 0.001$ level; ns non-significant.

Consequently, it could be argued that deriving a pedotransfer function to estimate pHBC from routinely determined soil attributes (i.e. clay and organic carbon content) over a wide range of agroecological zones would be problematic where there are soils of mixed mineralogy. Gillman and Sumpter (1986) observed that in soils of the wet tropics exhibiting variable charge characteristics, approximately 50% of the lime added to raise the pH of a soil suspensions from 4 to 6 was used in generating cation exchange capacity and not in directly increasing soil pH. Consequently, the variable charge nature of certain soils (i.e. ferrosols) will have a significant effect on pHBC. There are several potential errors that are inherent in the estimation of pHBC capacity that have been outlined by Helyar and Porter (1989). For example, the measurement of pH buffer capacity is in general derived from short-term laboratory incubation studies that do not take into account the slow pH buffering reactions (i.e. the dissolution of solid phases; biological processes) (Ridley *et al.* 1990). To overcome problems associated with short-term incubation studies, pH buffering capacity has been determined through field trials where pH changes have been monitored over a long period of time (Helyar *et al.* 1990). However, errors associated with incomplete dissolution or uneven mixing of lime has been suggested. In addition, acidification may occur on the unlimed treatments at different rates to that of limed treatments (Porter *et al.* 1995). Clearly the method used to determine pHBC will directly influence the ability to predict the quantity of lime required to neutralize acidity or to achieve a pre-determined pH level. It is generally agreed that short-term equilibration methods may underestimate buffering capacity and, therefore, lime requirement. However, pHBC is a dynamic characteristic and, therefore, any estimated measure is subject to ongoing change.

5.3 Influence of changed management of sugarcane on chemical properties

5.3.1 Methods and materials

Green Cane Trash Blanketing / Burnt trial

A long-term GCTB/burn trial was established in July 1990 at BSES Tully, north Queensland (17°59'S; 145°56' E). The climate of this region is tropical with a mean annual rainfall of approximately 4300 mm, concentrated in October-March. The treatments of interest in this evaluation were (a) GCTB: trash after mechanical harvesting retained on the soil surface with no soil cultivation and (b) Burnt: trash burnt before and after harvest and soil cultivated to a depth of 15 cm after harvest. The soil was classified as an acidic dystrophic yellow dermosol (Isbell 1996) or an oxic humitropept (Soil Survey Staff 1990). The trial was a completely randomized block design with three replications and was planted to Q124 with an annual application of 180 kg N/ha as urea. Seven 7 crops were harvested from the site before the trial was ploughed out and replanted. Soil samples were collected in October 1997, prior to ploughout, from 10 random points within a plot at depth intervals of 0-2, 2-5 and 5-10 cm and bulked to form composite depth samples.

Rotation trial

The second trial, also in the Tully area, was a rotation experiment established to investigate crop-management strategies to reduce the effects of yield decline in long-term sugarcane monocultures. Only three treatments of interest in this study are discussed in detail; Garside *et al.* (1999) give a comprehensive discussion of the trial. The three

treatments that were selected for this study were: (a) the long-term continuous GCTB; (b) pasture ley; and (c) a bare-fallow treatment. The site had been under continuous cane with minimal fallow breaks for the 20 years prior to the imposition of the trial. In November 1993, cane was removed from plots to be established to the pasture/legume ley consisting of a mixture of signal grass (*Brachiaria decumbens*) and pinto peanut (*Arachis pintoi*). No fertilizer was applied to the pasture during the study. The pasture was managed by periodic slashing, leaving the residue on the plots. Prior to each slashing, pasture growth was measured by quadrat sampling with dry matter production of tops being about 20 t/ha/yr. The bare fallow plot was maintained vegetation-free through periodic application of herbicide during the study. The continuous-cane plot was maintained over six consecutive crops. The soil was a yellow kandosol (Isbell 1996). The trial was a completely randomized block design with three replications.

Prior to the implementation of cane treatments in May 1994, soil samples (0-10 cm depth interval) were collected from all treatments at 30 individual points to form a composite sample for each plot. In August 2000, prior to the plough-out of all treatments, three random soil samples were collected from the 0-10 cm depth interval and bulked to form a composite sample for each plot.

Chemical analysis

Prior to chemical analysis, soils were air-dried and ground to pass through a 2-mm sieve before pH (1:5 water) was measured. Basic exchangeable cations were determined by atomic absorption spectrometry after replacement with 0.1M BaCl₂-NH₄Cl (Gillman and Sumpter 1986). Acidic cations were extracted with 1M KCl and the extractant titrated to pH 8.0 as described in Method 15G1 (Rayment and Higginson 1992). The effective cation exchange capacity (ECEC) was calculated as the sum of basic and acidic cations (Ca²⁺+Mg²⁺+K⁺+Na⁺+Al³⁺+H⁺). Organic carbon was determined by wet oxidation using the Walkley and Black (1934) method as described in Method 6A1 in Rayment and Higginson (1992).

Composite samples were produced for each of the depth intervals and treatments from both studies. Charge fingerprints, which are curves describing the total cation exchange capacity (CEC_T), base cation exchange capacity (CEC_B) and anion exchange capacity (AEC) across a range of pH values were determined on samples using the methodology described by Gillman and Sumpter (1986). In brief, soils were Ca saturated and brought to equilibrium in a 0.002M CaCl₂ matrix. Suspension pH was adjusted to six values ranging from approximately 4.0 to 7.5, and exchangeable Ca and Al displaced with NH₄NO₃. The Al content in solution was determined using the pyrocatechol-violet method (Bartlett *et al.* 1987) and Ca by atomic absorption spectrometry. Amounts adsorbed were calculated taking into account the amounts present in the entrained solutes. The CEC_B is operationally defined as the Ca adsorbed, and CEC_T as the Ca and Al adsorbed. Modifications to the method of Gillman and Sumpter (1986) were made so that the amount of acid or base added to achieve the desired pH range was quantified. This effectively allowed an estimation of the pH buffering capacity (pHBC) to be made on the composite samples. Dr M. Braunack, formerly of BSES, undertook soil bulk density measurements at each of the sites independently using the standard methodology of Blake and Hartge (1986).

To assess the possible contribution of particular functional organic groups to charge generation, we carried out carbon fractionation on the above composite samples using 33mM and 333mM KMnO_4 to discriminate between the oxidisability of the soil organic C (Blair *et al.* 1995). Conteh *et al.* (1999) showed that differential ease of oxidation was correlated with the proportional amounts of various functional organic groups in the soil organic matter. Total organic C was determined by dry combustion at 1300°C in a Leco carbon analyzer.

Table 6 Selected soil chemical properties after 7 years of treatments from a long-term trash retention/burnt cane trial established at Tully, north Queensland. Values in parentheses are the SE of the mean

Parameter	Treatment	Depth (cm)		
		0-2	2-5	5-10
pH _w	GCTB	5.11 (0.01)	5.13 (0.02)	5.26 (0.03)
	Burnt	5.33 (0.01)	5.33 (0.01)	5.41(0.07)
Ca (cmol _c /kg)	GCTB	2.49 (0.31)	1.96 (0.26)	1.88 (0.26)
	Burnt	1.86 (0.40)	1.77 (0.24)	1.70 (0.49)
Mg (cmol _c /kg)	GCTB	1.03 (0.08)	0.77 (0.02)	0.60 (0.02)
	Burnt	0.76 (0.08)	0.66 (0.00)	0.51 (0.06)
K (cmol _c /kg)	GCTB	0.39 (0.02)	0.19 (0.01)	0.10 (0.01)
	Burnt	0.30 (0.01)	0.20 (0.05)	0.14 (0.03)
Na (cmol _c /kg)	GCTB	0.11 (0.01)	0.09 (0.01)	0.09 (0.01)
	Burnt	0.12 (0.001)	0.09 (0.01)	0.07 (0.01)
Exchangeable acidity (cmol _c /kg)	GCTB	1.31 (0.23)	1.41 (0.25)	1.44 (0.28)
	Burnt	0.91 (0.06)	1.01 (0.01)	1.06 (0.16)
W-B organic C (g/kg)	GCTB	25.0 (1.6)	15.6 (0.4)	12.2 (0.4)
	Burnt	13.6 (2.1)	12.8 (0.6)	11.6 (1.1)
Total organic C (g/kg)	GCTB	30.0	16.4	12.5
	Burnt	14.7	13.2	12.3
33mM KMnO_4 oxidisable C (C1) (g/kg)	GCTB	3.04	1.21	0.71
	Burnt	0.93	0.80	0.67
333mM KMnO_4 oxidisable C (C3) (g/kg)	GCTB	6.74	3.95	1.86
	Burnt	2.43	2.02	2.28
pH buffer capacity (cmol H ⁺ /kg.pH unit)	GCTB	2.79	2.22	2.03
	Burnt	1.92	1.97	2.01
ECEC (cmol _c /kg)	GCTB	5.32 (0.30)	4.42 (0.14)	4.11 (0.06)
	Burnt	3.95 (0.54)	3.74 (0.20)	3.48 (0.38)
Bulk density (g/cm ³)	GCTB	1.13	1.28	1.25
	Burnt	1.17	1.24	1.26

5.3.2 Results and discussion

Changes in soil organic carbon, pH buffer capacity and exchangeable cations

GCTB/Burnt trial: Significant decreases in soil pH were recorded over the depth interval 0-5 cm under the trash retention system (Table 6). The largest difference in soil pH between the two management systems occurred in the 0-2 cm depth interval where a difference in pH of 0.22 units was recorded over the period of the trial (7 years). This pH decrease is to be expected, since this is the zone of greatest biological activity where the processes of mineralization and, hence, acid generation are at their greatest. There was a trend for increases in exchangeable Ca^{2+} , Mg^{2+} , K^+ and ECEC in the surface 0-2 cm depth under the GCTB treatments, significant ($P < 0.05$) only in the case of K (Table 6).

However, these differences diminished with depth. Associated with the decrease in pH, there was a corresponding increase in exchangeable acidity under the GCTB system when compared to the burnt (Table 6). Soil organic carbon levels in the top 0-2 cm increased substantially under the GCTB system, but this increase did not persist with depth. The amount of total soil organic carbon in the 10-cm depth interval of the GCTB system was equivalent to 20.9 t/ha whilst that under the burnt regime amounted to 16.1 t/ha. Consequently, a total of 4.8 t/ha of total organic carbon accumulated under the GCTB system over 7 years.

The pH buffering capacity of soils was highest in the 0-2 cm depth interval under the GCTB system and declined with depth until it was the same as that for the burnt treatment at 5-10 cm (Table 6). This increase in buffering capacity can be attributed to elevated soil organic carbon levels in the GCTB system, as there was a significant ($P < 0.05$) positive correlation between total organic C and pH buffer capacity (Figure 4). Using the pH buffer capacity of the GCTB treatment for each of the depth intervals, the net proton input in the GCTB treatment relative to the burnt was estimated to be 0.71 kmol/ha/yr using equation (2) (Helyar and Porter 1989).

$$\text{NAAR} = [[(\text{pH}_B \times \text{pH}_{\text{BCB}}) - (\text{pH}_T \times \text{pH}_{\text{BCT}})] \times \text{BD} \times \text{V}] / \text{T} \quad (2)$$

where the subscripts B and T refer to burn and trash retained production systems; pH_{BC} is the pH buffering capacity for each of the production systems ($\text{kmol H}^+/\text{kg.pH unit}$); BD is soil bulk density (kg/m^3); V is the soil volume in the depth interval under consideration (m^3/ha); and T is time (years) since the establishment of the trial.

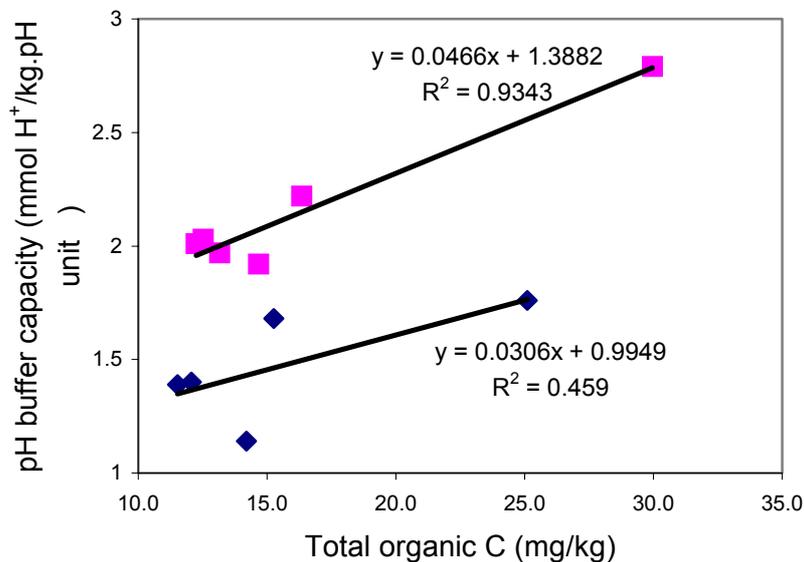


Figure 4 Relationship between pH buffer capacity and total organic C for the GCTB/Burnt Trial (■) and the Rotation Trial (◆)

Table 7 Changes in soil chemical properties (0-10 cm) associated with the conversion from continuous cane to a grass ley system 6 years after imposition of treatments at Feluga, north Queensland. Values in parentheses are the SE of the mean

Parameter	Treatment	Year	
		1994	2000
pH _w	Cont. cane	6.55 (0.19)	6.03 (0.21)
	Ley	6.40 (0.41)	5.46 (0.23)
	Fallow	6.03 (0.13)	5.76 (0.09)
Ca (cmol _c /kg)	Cont. cane	3.53 (0.54)	1.55 (0.14)
	Ley	3.26 (0.72)	1.83 (0.30)
	Fallow	2.15 (0.11)	1.77 (0.37)
Mg (cmol _c /kg)	Cont. cane	0.54 (0.06)	0.34 (0.01)
	Ley	0.43 (0.05)	0.37 (0.01)
	Fallow	0.34 (0.01)	0.15 (0.01)
K (cmol _c /kg)	Cont. cane	0.25 (0.02)	0.21 (0.02)
	Ley	0.22 (0.01)	0.24 (0.01)
	Fallow	0.17 (0.02)	0.14 (0.01)
Na (cmol _c /kg)	Cont. cane	0.03	0.08 (0.02)
	Ley	0.04 (0.04)	0.10 (0.01)
	Fallow	0.04	0.07 (0.01)
Exchangeable acidity (cmol _c /kg)	Cont. cane	0.07 (0.01)	0.09 (0.01)
	Ley	0.13 (0.05)	0.24 (0.04)
	Fallow	0.20 (0.05)	0.26 (0.20)
W-B organic C (g/kg)	Cont. cane	12.6 (0.04)	7.7 (0.5)
	Ley	12.6 (0.12)	14.3 (0.4)
	Fallow	12.3 (0.06)	7.9 (0.1)
Total organic C (g/kg)	Cont. cane	nd	14.2
	Ley	nd	25.1
	Fallow	nd	11.5
33mM KMnO ₄ oxidisable C (C1) (g/kg)	Cont. cane	nd	1.07
	Ley	nd	2.12
	Fallow	nd	0.68
333mM KMnO ₄ oxidisable C (C3) (g/kg)	Cont. cane	nd	2.21
	Ley	nd	5.56
	Fallow	nd	1.41
pH buffer capacity (cmol H ⁺ /kg.pH unit)	Cont. cane	nd	1.14
	Ley	nd	1.76
	Fallow	nd	1.39
ECEC (cmol _c /kg)	Cont. cane	4.43 (0.61)	2.27 (0.05)
	Ley	4.10 (0.68)	2.79 (0.20)
	Fallow	2.89 (0.16)	2.57 (0.22)
Bulk density (g/cm ³)	Cont. cane	nd	1.40 (0.04)
	Ley	nd	1.38 (0.04)
	Fallow	nd	1.39 (0.03)

Rotation trial: Table 7 gives changes in selected soil chemical properties for the 0-10 cm depth intervals at the start and end of the study. At the end of the study (2000), soil pH was significantly higher ($P < 0.05$) under the continuous cane treatment at 0-10 cm when compared to the grass ley (Table 7). Differences in pH were not evident at the 10-20 cm depth interval (data not presented).

In exchangeable basic cations there were no significant differences between treatments with depth (data not presented). The continuous cane treatment significantly ($P < 0.05$) decreased exchangeable Ca and Mg over the 6-year period, whereas exchangeable K levels did not change (Table 7). Pasture ley for 6 years (with return of all harvested material to the plots) did not cause any changes in the levels of exchangeable cations. After 6 years, exchangeable acidity was significantly ($P < 0.05$) higher under the pasture ley when compared to the continuous cane.

Total soil organic carbon increased from 19.9 t/ha under the continuous cane to 34.6 t/ha under the pasture ley over the 0-10 cm depth interval, indicating a 14.7 t/ha difference in total organic C between the systems over the 6-year period. Associated with the increase in organic C under the grass ley system there was a corresponding increase in the pH buffer capacity (Table 7).

Assuming that the pH buffer capacity over the intervening 6-year period did not change under a particular management system, the net acid inputs between 1994 and 2000 for the continuous cane, grass ley and fallow systems were 1.38, 3.81 and 0.87 kmol/ha/yr, respectively. Clearly, the greater net proton input under the grass ley system can in part be explained by the increase in the organic carbon content of the soil (Helyar and Porter 1989).

Surface charge characteristics

The concept of surface charge fingerprinting as a means of describing the electrochemical characteristics of a soil was first described by Gillman and Sumpter (1986). The approach distinguishes that portion of the cation exchange capacity (CEC) that retains basic cations, and predicts changes in CEC as soil pH and ionic strength are varied. To assess this, we determined charge fingerprints on a composite sample for treatments collected from the 0-2, 2-5 and 5-10 cm depth intervals in the case of the GCTB/Burnt Trial and the 0-10 and 10-20 cm depth intervals for the Rotation Trial (Figures 5 and 6, respectively). The CEC_B , as depicted in Figures 5 and 6, is a measure of the maximum amount of exchangeable basic cations that can be retained in exchangeable form, and is, therefore, of agricultural significance. In contrast, the total cation exchange capacity (CEC_T) is an estimate of the charge associated with both acidic and basic cations, and only becomes significant at pH values of < 5.5 . It is clearly evident that there are substantial differences in the charge characteristics associated with different management systems. In both of the trials under review, there is a vertical upward shift in the charge curve that is more pronounced in the surface horizons but diminishes with depth. The impact of GCTB on charge characteristics is confined to the 0-5 cm depth interval and can be directly attributed to the increased soil organic matter content (Figure 7). Similarly, the effects of the grass ley on charge characteristics were confined to the 0-10 cm depth.

At pH 5.5 there is a convergence in the two charge curves, indicating that the CEC is base saturated. This is of agronomic significance, since it clearly shows that by increasing the

pH to this value any potential problems associated with acidic cations (i.e. Al^{3+} and H^+) would be eliminated and the base cation complement on the exchange complex maximised.

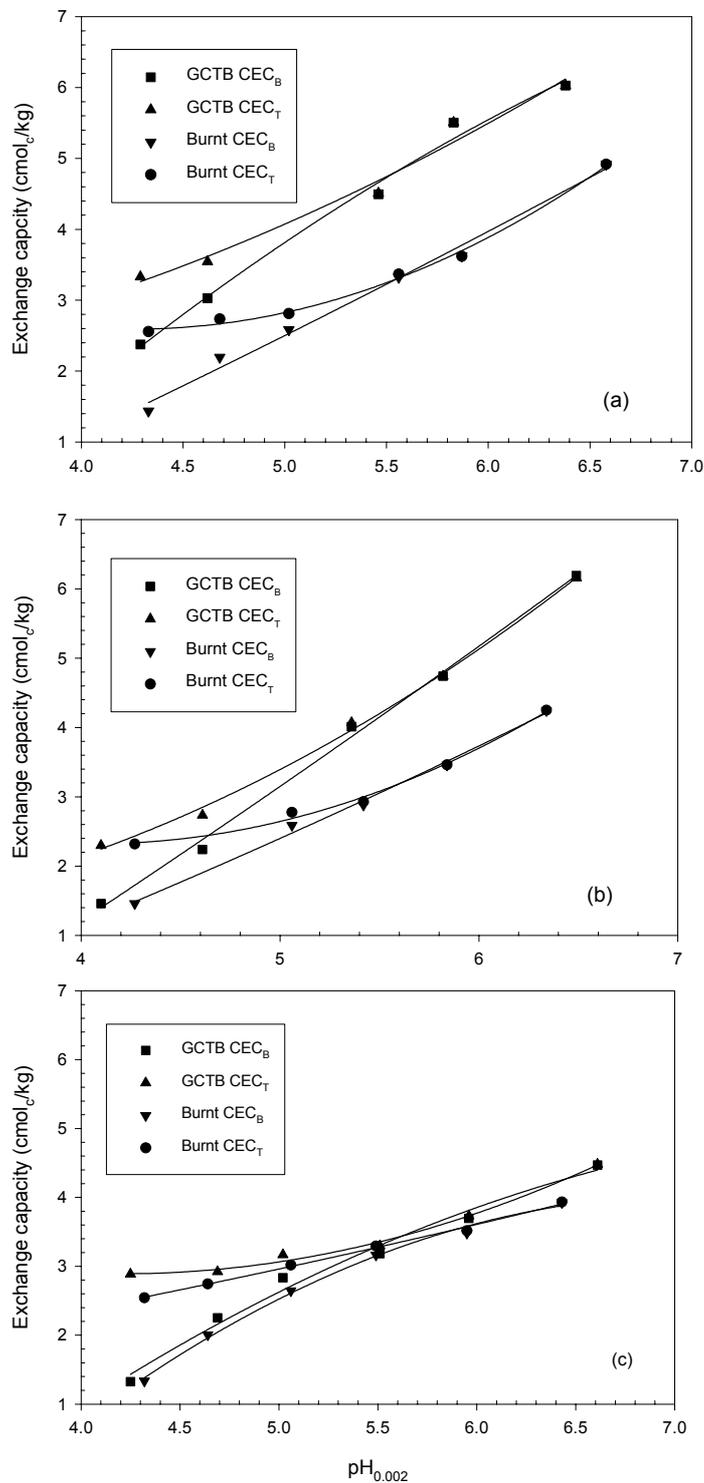


Figure 5 Surface charge fingerprints of the (a) 0-2 cm, (b) 2-5 cm and (c) 5-10 cm depth intervals under a long-term GCTB and burnt system. CEC_T is represented by dotted lines and CEC_B by solid lines

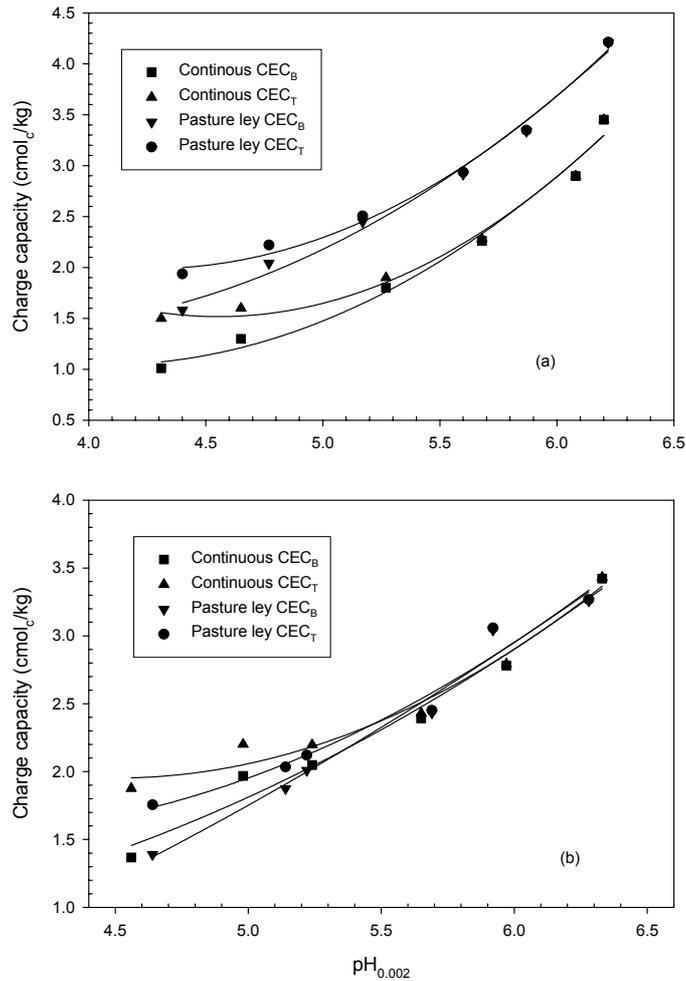


Figure 6 Changes in surface charge characteristics associated with the establishment of a pasture ley system in the (a) 0-10 cm and (b) 10-20 cm depth intervals

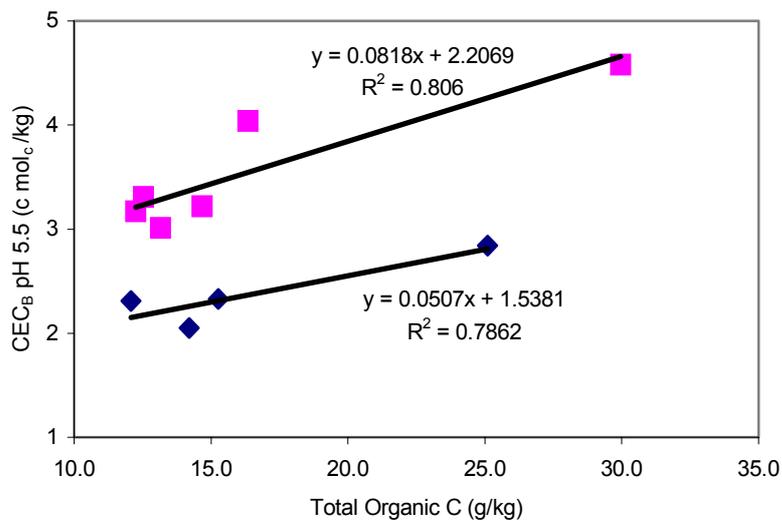


Figure 7 Relationships between CEC_B at pH 5.5 and total organic C in the 0-10 cm layer for the GCTB/Burnt Trial (■) and the Rotation Trial (◆)

To quantify the degree of charge generation associated with changed management, CEC values should be compared at the same pH (i.e. pH 5.5). For the 0-2, and 2 - 5 cm depth intervals the CEC increases between the burnt and GCTB systems were 1.50 and 1.11 cmol_e/kg , respectively (Figure 5). However, the increase in CEC between the continuous cane and ley pasture system over the 0-10 cm was 0.75 cmol_e/kg (Figure 6). By taking a weighted average of the CEC over a 10 cm depth interval, the increase in CEC between the burnt and GCTB treatments reduces to 0.67 cmol_e/kg .

The increase in CEC associated with the changed management can be attributed to changes in soil organic carbon, as there was a significant ($P < 0.05$) positive correlation between total organic C and the interpolated CEC_B at pH 5.5 (Figure 7). Data from the two sites lie on separate regression lines, with the GCTB/Burnt Trial having a higher CEC_B at a given total organic C content than the Rotation Trial. Extrapolation of the regression lines to zero total organic C indicates that the charge associated with clay minerals in the two soils equates to 2.2 cmol_e/kg and 1.5 cmol_e/kg for the GCTB/Burnt Trial and the Rotation Trial, respectively. This difference is attributed to the different clay contents of the 0-10 cm layers of the two sites, 36.8 % and 20.2 %, respectively.

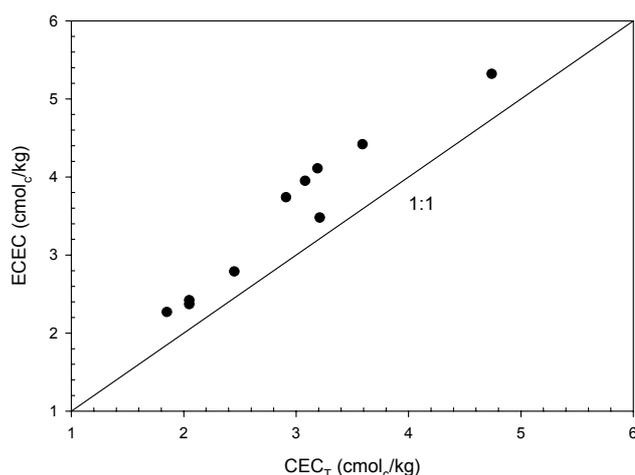


Figure 8 Relationship between effective cation exchange capacity (ECEC) and the total cation exchange capacity (CEC_T) as determined from the surface charge fingerprint for all sites and depth intervals at soil pH

An important characteristic of the charge fingerprint is that it allows an estimation of CEC_B and CEC_T at soil pH. Consequently, if the basic and acidic cations removed by the $\text{BaCl}_2/\text{NH}_4\text{Cl}$ and KCl extractants, respectively, are all exchangeable cations, then their sum (the ECEC) should be equal to CEC_T at soil pH, within the limits of the experimental error associated with these estimations. A comparison of these two parameters for all data from both trial sites clearly shows that in all cases the ECEC is larger than CEC_T at soil pH, indicating that greater amounts of cations were extracted than could be accounted for by the CEC_T (Figure 8). The magnitude of these differences ranged from 9-31% of the actual measured CEC. This would suggest that a component of the cation complement that is extracted by 0.1M $\text{BaCl}_2/\text{NH}_4\text{Cl}$ is not held on the exchange complex and is,

therefore, potentially subject to leaching. A similar discrepancy between CEC_T and ECEC was observed by Noble *et al.* (2000) on soils sampled during the dry season. We speculate that, since these samples were taken during the drier part of the year, mineralisation of organic matter may have occurred, resulting in an accumulation of soluble cations in the surface horizons. This clearly shows one of the advantages of constructing charge fingerprints since it highlights that a component of the extractable cations are in a soluble form and, therefore, vulnerable to leaching.

Organic carbon fractionation

Organic C oxidisable by 33mM $KMnO_4$ (C1) or 333mM $KMnO_4$ (C3) was highly linearly correlated with total organic C across both sites and all treatments [$r^2 = 0.98$ and 0.95 , respectively ($P < 0.001$)]. This suggests that the proportions of different functional organic groups were not greatly affected by burning, trash retention or pasture. The similarity in composition of the soil organic C between sites and across treatments is also evident in that the slopes of the linear regressions between CEC_B and total organic C (TOC), and between pH buffer capacity (pHBC) and TOC for the two sites are not significantly ($P < 0.05$) different (Table 8).

Table 8 Linear regression parameters for relationships between CEC_B at pH 5.5 and total organic C and between pH buffer capacity and total organic C. Standard errors are in parentheses

Relationship	Slope	Intercept	r^2
CEC_B versus total organic C			
GCTB/Burnt Trial	0.082 (0.020)	2.21 (0.35)	0.81
Rotation Trial	0.059 (0.019)	1.54 (0.32)	0.79
pHBC versus total organic C			
GCTB/Burnt Trial	0.047 (0.006)	1.388 (0.109)	0.93
Rotation Trial	0.033 (0.026)	0.948 (0.446)	0.46

5.4 Assessment of the effect of water-soluble organic components leached from plant materials on the cation mobilisation in acid light-textured soils

5.4.1 Methods and materials

Plant residues

Plant residues used in this study and region from where they were collected are given in Table 9. The species were diverse, in that they represent different plant orders. In this respect the grasses (monocots) are represented by sugarcane and *Urochloa mosambicensis*; leguminous dicots represented by *Stylosanthes* collected from the semi-arid tropics; and freshly fallen leaf-litter material collected from the tree species *Melia azedarach*. *Melia azedarach* was used as a benchmark species, since it has been recognised as having a high ash alkalinity (Noble *et al.* 1996). Samples were dried at 65°C for 48 h in a forced-draft oven and ground to pass through a 2-mm sieve.

Table 9 Type and collection location of plant residues used in leaching studies

Plant material	Collection location
Sugarcane trash	Herbert River District, north Queensland
White cedar (<i>Melia azedarach</i>) senesced leaf material	Canberra, ACT
<i>Stylosanthes</i> (Seca) whole-plant material	Lansdown, north Queensland
<i>Urochloa mosambicensis</i>	Lansdown, north Queensland

Preparation of leaching columns

The surface 0-15 cm of an acid light-textured sand collected was collected from the Herbert River district, north Queensland. Particle size analysis showed that soil was dominated by sand and was classified as having a sandy loam texture. The site had supported sugarcane for the past 30 years. Soil was air dried and sieved to pass through a 2-mm mesh. The leaching columns were 15-cm long perspex tubes with an internal diameter of 2.5 cm. Columns were packed to a depth of 12 cm at a uniform bulk density of 1.48 g/cm³. Each column contained 87 g of soil and a total pore volume of 18.61 cm³. Prior to packing the column, a 2.5 cm Millipore GF filter pad was placed at the bottom of the column to prevent soil from falling out of the column. Once the column was packed, a Millipore GF filter was placed on the soil surface.

Preparation of the litter material

A mixture of litter and acid washed sand was placed on the top of the Millipore GF filter. This mix consisted of 1.227 g (equivalent to 50 t/ha dry matter) of litter and 5 g of acid washed sands thoroughly mixed. A third Millipore GF filter was placed on top of the surface of the litter/sand mix to distribute the distilled water evenly over the surface of the soil within the column and prevent surface impact. Columns were leached with distilled water over an 8-h period with a total of 5.4 pore volumes. After this leaching phase, the surface placed trash/sand mixture was carefully removed from the surface of the column to expose the GF filter separating the soil from the litter/sand mixture. A further 20 mL of distilled water was passed through the column. The effluent emanating from the bottom of the column was collected and the pH and EC measured at the completion of the leaching phase. The above procedure was undertaken for each of the litter materials and was replicated three times. In addition, a sample of the trash/sand mixture was leached with 5.4 pore volumes of distilled water and the leachate collected. This represented the composition of the leaching emanating from the trash/sand mixture prior to passing through the soil column.

An additional litter treatment was prepared with cane trash, where the sand-trash mixture was treated with KOH. This treatment was undertaken to assess the effect of applying alkaline materials on the composition of the leachate emanating from the trash. The trash/sand mixture was prepared by taking 6.14 g of sugarcane trash and mixing it with 25 g acid washed sand. 0.30 g KOH was mixed with 20 mL of distilled water, added to the trash/sand, thoroughly mixed and allowed to equilibrate for 16 h. The mixture was then oven dried at 40°C over 24 h and the above leaching procedure undertaken. As a control treatment, columns were prepared with only soil and leached as described previous.

Soil and leachate analysis

At the completion of the leaching phase, the columns were allowed to drain free and the soil gently blown out of the columns. The columns were sectioned into 2-cm sections from the top to the bottom of the column. Soils were dried at 40°C in a forced-draft oven for 48 h and passed through a 2 mm sieve.

Soil pH and EC were measured in a 1:5 soil:water solution matrix. Exchangeable cations (Ca^{2+} , Mg^{2+} , K^{+}) were extracted using 0.1M $\text{BaCl}_2/\text{NH}_4\text{Cl}$ (Gillman and Sumpter 1986) and determined by atomic absorption spectroscopy. Exchangeable Al^{3+} was determined on the leachates using the pyrocatechol-violet method (Bartlett *et al.* 1987).

The leachate extract was stored at 4°C until analysis. The dissolved organic carbon was determined using the modified Mebius method of Yeomans and Bremner (1988). The cation concentration was determined by atomic absorption spectroscopy.

5.4.2 Results and discussion

Current practices of GCTB management dictate that a large volume of trash is returned to the soil surface after harvest. This trash blanket, which is allowed to breakdown over the course of the subsequent growing season, releases significant amounts of nutrients and organic carbon back into the soil. Contrasting this, in burnt-cane production systems there is a significant quantity of nutrients release on combustion that is effectively readily available for possible uptake by the crop or lost through leaching and runoff. It is important to quantifying the impact of these different management strategies on soil chemical properties. In undertaking this assessment, sugarcane trash should be compared with other litter materials that differ in chemical composition. Hence, litter materials that have contrasting chemical composition were evaluated.

Effect of litter/trash material on soil properties and leachate composition

Soil pH: As expected, there was no significant change in soil pH with depth in the Control treatment with leaching (Table 10). However, with the addition of litter material to the surface of columns and subsequent leaching, significant ($P < 0.05$) increases in pH were evident with depth down the column, although the largest increases were observed in the surface 0-6 cm (Table 10). The unamended cane litter had a significant effect on soil pH when compared with the Control, with an effect on pH to a depth of 6 cm. However, the greatest increase in pH with depth was achieved where KOH was mixed into the sugarcane trash, emulating the potential role of surface treating trash blankets with alkaline materials to change the characteristics of the litter. Soil pH was increased from 5.50 to 6.54 in the surface 0-2 cm in the cane trash and cane trash + KOH treatments respectively (Table 10). Contrastingly, *Melia*, *Stylosanthes* and *Urochloa* all resulted in significant increases in soil pH in the surface 0-2 cm, but these effects diminished with depth, with *Melia* showing the highest degree of persistence (Table 10). There appeared to be little difference in soil pH between the *Stylosanthes* litter collected from the soil surface and freshly harvested *Stylosanthes* plant material (Table 10).

By taking the mean pH of the litter treatments over the entire depth of the column and subtracting it from the mean of the Control, an estimate of the degree of shift in pH associated with the treatments can be achieved and used to compare between ashed and non-ashed treatments (see below). These differences in pH were 0.29, 0.81, 0.66, 0.38,

0.38 and 0.47 for the cane trash, cane trash +KOH, *Melia*, *Stylosanthes* litter, *Stylosanthes* fresh and *Urochloa*, respectively.

Table 10 Changes in soil pH_w with depth after leaching with selected plant litter materials

Treatment	Depth (cm)					
	0-2	2-4	4-6	6-8	8-10	10-12
Control	4.96	5.01	5.01	5.02	4.95	4.99
Cane trash	5.50	5.34	5.26	5.22	5.21	5.19
Cane trash + KOH	6.54	6.12	5.97	5.67	5.35	5.16
<i>Melia azedarach</i> leaf litter	5.61	5.62	5.72	5.50	5.28	5.21
<i>Stylosanthes</i> litter collected from soil surface	5.62	5.43	5.32	5.32	5.28	5.27
<i>Stylosanthes</i> fresh material	5.68	5.45	5.29	5.22	5.17	5.13
<i>Urochloa mosambicensis</i> fresh material	5.97	5.72	5.49	5.29	5.15	5.15
LSD (0.05)	0.13					

Exchangeable cations: Mean exchangeable Ca²⁺ content in the Control treatment was 0.36 cmol_c/kg (Table 11). Contrastingly, the greatest increase in Ca²⁺ on the exchange complex was observed in the treatment leached with extracts emanating from the *Melia* litter, where there was an average of 1.30 cmol_c/kg of Ca²⁺ down the entire profile. In previous leaching column studies using *Melia azedarach* litter material, similar changes in exchangeable Ca²⁺ levels have been observed, this being attributed to the high Ca content in that litter (Noble *et al.* 1996). There were little, if any, significant changes in Ca²⁺ content on the exchange complex with leaching in the other litters.

Table 11 Changes in exchangeable Ca²⁺ with depth after leaching with selected plant litter materials

Treatment	Depth (cm)					
	0-2	2-4	4-6	6-8	8-10	10-12
Control	0.36	0.36	0.34	0.36	0.35	0.38
Cane trash	0.32	0.18	0.19	0.21	0.22	0.22
Cane trash + KOH	0.25	0.24	0.27	0.32	0.36	0.39
<i>Melia azedarach</i> leaf litter	1.81	1.49	1.38	1.25	0.98	0.90
<i>Stylosanthes</i> litter collected from soil surface	0.47	0.24	0.25	0.25	0.26	0.29
<i>Stylosanthes</i> fresh material	0.42	0.23	0.29	0.27	0.26	0.25
<i>Urochloa mosambicensis</i> fresh material	0.36	0.39	0.52	0.64	0.63	0.71
LSD (0.05)	0.25					

With the exception of the cane trash + KOH treatment, exchangeable Mg²⁺ in the 0-2 cm depth significantly ($P < 0.05$) increased with leaching (Table 12). The largest increase occurred in the *Urochloa*, followed by the cane-trash treatments. In the *Melia* treatment, significant changes in Mg²⁺ were evident down the entire depth of the leaching column; this was probably associated with the elevated Mg²⁺ concentration in leachate (see

discussion below). Significant declines in the Mg^{2+} were observed in the cane trash + KOH treatment associated with elevated K^+ concentrations in the leachate.

Table 12 Changes in exchangeable Mg^{2+} with depth after leaching with selected plant litter materials

Treatment	Depth (cm)					
	0-2	2-4	4-6	6-8	8-10	10-12
Control	0.14	0.13	0.13	0.13	0.14	0.15
Cane trash	0.43	0.15	0.12	0.13	0.13	0.14
Cane trash + KOH	0.10	0.05	0.05	0.07	0.09	0.11
<i>Melia azedarach</i> leaf litter	0.23	0.21	0.23	0.24	0.22	0.23
<i>Stylosanthes</i> litter collected from soil surface	0.21	0.11	0.13	0.13	0.14	0.16
<i>Stylosanthes</i> fresh material	0.22	0.11	0.13	0.14	0.14	0.15
<i>Urochloa mosambicensis</i> fresh material	0.63	0.13	0.10	0.14	0.15	0.18
LSD (0.05)	0.04					

The addition of the litter to the soil surface in the columns resulted in significant increases in exchangeable K^+ to depth (Table 13). As expected, the greatest increase was observed in those treatments receiving cane trash + KOH, where additional K^+ was supplied to the leachate (Table 13).

Table 13 Changes in exchangeable K^+ with depth after leaching with selected plant litter materials

Treatment	Depth (cm)					
	0-2	2-4	4-6	6-8	8-10	10-12
Control	0.13	0.13	0.14	0.14	0.14	0.16
Cane trash	0.37	0.39	0.33	0.26	0.19	0.18
Cane trash + KOH	1.82	1.21	1.06	0.77	0.54	0.39
<i>Melia azedarach</i> leaf litter	0.34	0.46	0.50	0.43	0.44	0.41
<i>Stylosanthes</i> litter collected from soil surface	0.36	0.33	0.26	0.19	0.18	0.20
<i>Stylosanthes</i> fresh material	0.56	0.36	0.26	0.21	0.19	0.18
<i>Urochloa mosambicensis</i> fresh material	0.86	0.75	0.47	0.29	0.18	0.17
LSD (0.05)	0.12					

The sum of exchangeable bases is presented in Table 14; there has been a significant increase in the sum of basic cations over the control after leaching. Whilst exchangeable Al^{3+} was not determined on samples, part of the increase can be attributed to the displacement of Al^{3+} from the exchange complex through complexation with metallo-organic ligands and through precipitation within the column due to an increase in pH. In addition, soluble carbon that is retained in the column could potentially contribute to increased CEC, although as discussed below there was little retention within the column.

Table 14 Changes in sum of bases (sum Ca²⁺, Mg²⁺, K⁺) with depth after leaching with selected plant litter materials

Treatment	Depth (cm)					
	0-2	2-4	4-6	6-8	8-10	10-12
Control	0.65	0.66	0.64	0.68	0.66	0.73
Cane trash	1.13	0.73	0.65	0.61	0.55	0.56
Cane trash + KOH	2.20	1.52	1.41	1.19	1.02	0.92
<i>Melia azedarach</i> leaf litter	2.40	2.18	2.12	1.96	1.65	1.57
<i>Stylosanthes</i> litter collected from soil surface	1.08	0.73	0.69	0.61	0.64	0.71
<i>Stylosanthes</i> fresh material	1.25	0.76	0.75	0.69	0.65	0.63
<i>Urochloa mosambicensis</i> fresh material	1.89	1.31	1.15	1.11	1.00	1.12
LSD (0.05)	0.31					

Leachate characteristics: EC, Ca, Mg and K levels were all significantly reduced in the leachate after passing through the column in all treatments other than the cane trash + KOH treatment (Table 15), suggesting that retention of these cations within the column had occurred. In contrast, in the treatment receiving KOH, the concentrations of Ca²⁺ and Mg²⁺ increased in the leachate as evidenced by the decrease on the exchange complex (Tables 11 and 12), whilst the K⁺ concentration declined. pH declined in all leachates after passing through the column with a concomitant increase in Al (Table 15).

Table 15 Leachate and extract composition before and after passing through a soil column. Values in parenthesis represent the SE of the mean

Treatment	EC		pH		OC		Al	
	Extract	Leachate	Extract	Leachate	Extract	Leachate	Extract	Leachate
	(mS/cm)				(%)		(mg/L)	
Control	na	0.12 (0.02)	na	5.17 (0.26)	na	0.02 (0.01)	Na	0.11 (0.01)
Cane trash	0.39	0.33 (0.01)	5.71	4.69 (0.04)	0.24	0.18 (0.01)	0.16	0.30 (0.01)
Cane trash + KOH	1.43	0.60 (0.05)	9.63	4.40 (0.04)	0.34	0.25 (0.02)	0.10	0.72 (0.15)
<i>Melia azedarach</i> leaf litter	1.25	0.75 (0.02)	5.52	4.43 (0.06)	0.67	0.56 (0.01)	0.15	6.33 (0.69)
<i>Stylosanthes</i> litter collected from soil surface	0.21	0.21 (0.01)	6.68	4.68 (0.02)	0.08	0.06 (0.01)	0.09	0.21 (0.01)
<i>Stylosanthes</i> fresh material	na	0.19 (0.01)	Na	4.75 (0.02)	0.06	0.12 (0.01)	0.09	0.23 (0.02)
<i>Urochloa mosambicensis</i> fresh material	0.76	0.57 (0.01)	6.06	4.79 (0.01)	0.24	0.18 (0.00)	0.16	0.65 (0.07)

Treatment	Ca		Mg		K	
	Extract	Leachate	Extract	Leachate	Extract	Leachate
	(mg/L)		(mg/L)		(mg/L)	
Control	na	0.12 (0.05)	na	0.07 (0.01)	na	0.13 (0.02)
Cane trash	0.36	0.42 (0.02)	1.09	0.59 (0.01)	1.79	0.45 (0.01)
Cane trash + KOH	0.06	1.14 (0.46)	0.10	0.94 (0.05)	8.02	1.49 (0.12)
<i>Melia azedarach</i> leaf litter	7.23	1.42 (0.24)	2.56	1.69 (0.05)	4.23	1.83 (0.27)
<i>Stylosanthes</i> litter collected from soil surface	0.52	0.27 (0.03)	0.38	0.30 (0.01)	0.92	0.33 (0.01)
<i>Stylosanthes</i> fresh material	0.12	0.23 (0.02)	0.10	0.25 (0.01)	0.84	0.31 (0.00)
<i>Urochloa mosambicensis</i> fresh material	0.03	1.17(0.09)	0.59	0.58(0.02)	3.21	0.84(0.56)

There were relatively little if any changes in OC concentrations in the leachate after passing through the soil column (Table 15), suggesting little retention within the column. However, a significant relationship between the OC content in the extract prior to passing through the column and the amount of Al leached from the soil (Figure 9) clearly indicated the role of dissolved organic carbon in the formation of organo-metal complexes. It is also evident that the litter extracts differ in their ability to complex Al, since the OC concentrations in the *Urochloa* and cane-trash treatments were the same in the extract, but the concentrations of Al in the leachate were 0.65 and 0.30 mg/L, respectively (Table 15).

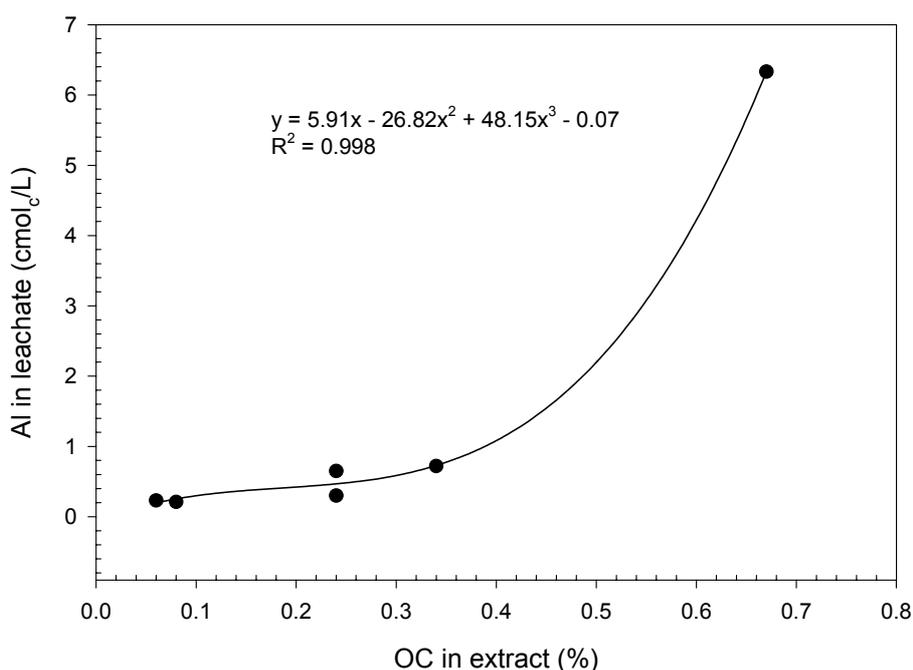


Figure 9 Relationship between OC in the leaching extract of litter materials prior to passing through a soil column and the Al³⁺ concentration after passing through the column

Effect of ashed litter/trash on soil properties and leachate composition

Soil pH: Soil pH was significantly increased in the surface 0-2 cm after leaching with the ashed extract (Table 16). The greatest increase in soil pH was in the ashed *Melia*, followed by the ashed cane-trash treatments (Table 16). An evaluation of the Δ pH resulted in changes in pH from the control of 0.38, 0.37, 0.22, 0.33 and 0.42 for the cane trash, *Melia*, *Stylosanthes* litter, *Stylosanthes* fresh and *Urochloa*, respectively. When these values are compared with the unashed treatments, the *Melia* treatment has undergone a dramatic decline in effectiveness i.e. 0.66 to 0.37. This suggests that with the combustion of *Melia* litter significant alkalinity is lost. The lack of movement of alkalinity (pH increases) to depth within the column with ashing suggests that much of the inherent alkalinity that was in the litter is either lost on combustion and /or used to generate surface charge in the 0-2 cm depth interval (see below for discussion).

Table 16 Changes in soil pH with depth after leaching with selected plant ashed litter materials

Treatment	Depth (cm)					
	0-2	2-4	4-6	6-8	8-10	10-12
Control	5.24	5.24	5.24	5.24	5.24	5.24
Cane trash	6.42	5.73	5.52	5.45	5.31	5.34
<i>Melia azedarach</i> leaf litter	6.61	5.75	5.36	5.31	5.31	5.36
<i>Stylosanthes</i> litter collected from soil surface	5.62	5.57	5.40	5.39	5.38	5.40
<i>Stylosanthes</i> fresh material	6.14	5.59	5.43	5.41	5.40	5.43
<i>Urochloa mosambicensis</i> fresh material	6.41	5.81	5.55	5.47	5.38	5.35
LSD (0.05)	0.12					

Exchangeable cations: Changes in exchangeable Ca^{2+} and Mg^{2+} were confined to the surface 0-2 cm depth interval and, thereafter, there was little change (Tables 17 and 18). The greatest increases in these two elements were observed in those treatments under the *Melia* and *Stylosanthes* litter treatments (Table 17 and 18). In the unashed *Melia* treatment exchangeable Ca^{2+} and Mg^{2+} increased down the entire column, suggesting that the extract from this litter was more effectively in distributing cations down the soil column when compared to combusted litter material. Exchangeable K^+ followed the same trends Ca^{2+} and Mg^{2+} with changes being confined to the surface 0-2 cm and ashed cane trash having the greatest impact (Table 19). Associated with the increase in soil pH in the 0-2 cm depth interval, there was a corresponding increase in the sum of bases on the exchange complex (Table 20).

Table 17 Changes in exchangeable Ca^{2+} with depth after leaching with selected plant ashed litter materials

Treatment	Depth (cm)					
	0-2	2-4	4-6	6-8	8-10	10-12
Control	0.76	0.76	0.76	0.76	0.76	0.76
Cane trash	0.53	0.52	0.61	0.68	0.70	0.66
<i>Melia azedarach</i> leaf litter	1.20	0.73	0.69	0.67	0.76	0.67
<i>Stylosanthes</i> litter collected from soil surface	1.17	0.63	0.71	0.71	0.73	0.71
<i>Stylosanthes</i> fresh material	0.74	0.60	0.63	0.63	0.61	0.65
<i>Urochloa mosambicensis</i> fresh material	0.60	0.51	0.61	0.65	0.61	0.65
LSD (0.05)	0.12					

Table 18 Changes in exchangeable Mg²⁺ with depth after leaching with selected plant ashed litter materials

Treatment	Depth (cm)					
	0-2	2-4	4-6	6-8	8-10	10-12
Control	0.16	0.16	0.16	0.16	0.16	0.16
Cane trash	0.19	0.09	0.12	0.15	0.15	0.15
<i>Melia azedarach</i> leaf litter	0.63	0.12	0.13	0.13	0.14	0.14
<i>Stylosanthes</i> litter collected from soil surface	0.32	0.12	0.14	0.15	0.15	0.14
<i>Stylosanthes</i> fresh material	0.26	0.13	0.15	0.14	0.15	0.15
<i>Urochloa mosambicensis</i> fresh material	0.25	0.10	0.10	0.12	0.11	0.15
LSD (0.05)	0.01					

Table 19 Changes in exchangeable K⁺ with depth after leaching with selected plant ashed litter materials

Treatment	Depth (cm)					
	0-2	2-4	4-6	6-8	8-10	10-12
Control	0.13	0.13	0.13	0.13	0.13	0.13
Cane trash	1.16	0.31	0.19	0.15	0.14	0.15
<i>Melia azedarach</i> leaf litter	0.99	0.39	0.18	0.13	0.15	0.13
<i>Stylosanthes</i> litter collected from soil surface	0.21	0.15	0.14	0.14	0.14	0.14
<i>Stylosanthes</i> fresh material	0.73	0.16	0.13	0.13	0.13	0.13
<i>Urochloa mosambicensis</i> fresh material	0.96	0.35	0.29	0.21	0.15	0.14
LSD (0.05)	0.13					

Table 20 Changes in the sum of bases (Ca²⁺, Mg²⁺ and K⁺) with depth after leaching with selected plant ashed litter materials

Treatment	Depth (cm)					
	0-2	2-4	4-6	6-8	8-10	10-12
Control	1.65	1.65	1.65	1.65	1.65	1.65
Cane trash	1.96	1.15	1.25	1.35	1.34	1.26
<i>Melia azedarach</i> leaf litter	2.87	1.49	1.34	1.24	1.39	1.25
<i>Stylosanthes</i> litter collected from soil surface	1.78	1.11	1.29	1.33	1.34	1.32
<i>Stylosanthes</i> fresh material	1.82	1.11	1.16	1.13	1.11	1.18
<i>Urochloa mosambicensis</i> fresh material	1.98	1.21	1.32	1.25	1.08	1.18
LSD (0.05)	0.25					

Leachate characteristics: In general the concentration of Ca, Mg and K in the extracts was lower (Table 21) than that from the un-ashed samples (Table 15). For example, the concentration of Ca in the extract from the ashed and unashed *Melia* litter was 0.24 and 7.23 mg/L, respectively (Tables 15 and 21), suggesting that on combustion there is the potential for elements to be volatilized. However, it is more plausible that with combustion sparingly soluble oxides of these metals are produced that restrict the rate of dissolution. The OC content in the extracts was undetectable, suggesting OC was lost on combustion. Similarly, there were trace concentrations of Al³⁺ in the leachates clearly showing the efficacy of organo-metallic complexation in removing Al from the exchange complex.

Table 21 Leachate and extract composition from ashed litter/trash materials before and after passing through a soil column

Treatment	EC		pH		OC		Al		Ca		Mg		K	
	Ext	Lch	Ext	Lch	Ext	Lch	Ext	Lch	Ext	Lch	Ext	Lch	Ext	Lch
	(µS/cm)				(%)		(mg/L)		(mg/L)		(mg/L)		(mg/L)	
Control						0.024		0.11		0.12		0.07		0.13
Cane trash	273	258	9.54	5.09	0.00	0.010	0.00	0.01	0.05	0.56	0.06	0.16	1.53	0.11
<i>Melia azedarach</i> leaf litter	341	245	10.34	5.19	0.00	0.002	0.00	0.01	0.24	0.43	0.22	0.12	1.38	0.09
<i>Stylosanthes</i> litter collected from soil surface	139	179	9.22	5.41	0.00	0.012	0.01	0.01	0.61	0.28	0.19	0.07	0.13	0.07
<i>Stylosanthes</i> fresh material	209	209	9.59	5.21	0.01	0.012	0.01	0.01	0.31	0.34	0.14	0.09	0.76	0.07
<i>Urochloa mosambicensis</i> fresh material	424	387	9.73	5.01	0.00	0.008	0.04	0.01	0.25	0.85	0.15	0.36	2.14	0.14
LSD (0.05)						ns		0.01		0.08		0.02		0.03

Ext = Extract; Lch = Leachate
ns = not significant at P = 0.05.

5.5 Conclusions

Estimation of net proton addition rates under a continuous cane trash blanketed production system is achieved through an assessment of temporal changes in pHBC and pH. This implies measurements of these two soil attributes at a point prior to the imposition of treatments and, subsequently, at some point in the future. Within the current suite of long-term trash-retention versus burnt trials established at Harwood, Mackay and Tully this could not be undertaken as the soil analytical data were not available at the initiation of the study. Hence, we had to resort to an alternative set of studies within the YDJV where soil samples were collected prior to the establishment of treatments.

The increased net proton addition in the GCTB system relative to the burnt system was estimated to be 0.71 kmol H⁺/ha.year and the net proton additions for the continuous cane, grass ley and fallow systems were 1.38, 3.81, and 0.87 kmol H⁺/ha.year, respectively. Moody and Aitken (1997) estimated rates of acid addition under continuous sugarcane production systems to be about 3.2 kmol H⁺/ha/yr, which is about 2.5 times higher than the estimates that we made. These differences in net acidification rates may in part be attributed to methodology of measuring pHBC and the time over which the assessment was made. To put these rates of acidification in the context of other production systems in the tropics, a banana production system produces 34.2 kmol H⁺/ha/yr, whilst a nitrogen-fertilized tropical pasture produces 6.4 kmol H⁺/ha/yr (Moody and Aitken 1997). Clearly, sugarcane-production systems are not major generators of acid when compared to these other production systems. However, if acid accumulation associated with these sugarcane production systems are not addressed, potential long-term damage to the soil resource base will occur that will impact on the long-term sustainability and productivity of these production systems.

In moving from a burnt-cane production system to GCTB there appeared to be a slightly greater degree of acidification under the latter system. However, this statement should be treated with caution in that this effect was only observed at a single site (i.e. Tully trial).

In fact, the results from the leaching column study indicate that the water-soluble fraction that emanate from unashed cane trash are more effective in transferring alkalinity to depth than ashed trash. Moreover, treating the cane trash with KOH proved to be more effective in transferring alkalinity to depth, when compared to cane trash on its own. This outcome is significant in that it suggests that introducing a highly alkaline compound to the trash significantly alters the characteristics of the leachate, thereby having a positive impact on alkalinity transfer. This potentially could be a means of neutralizing annual acid generation and supplying K to the crop. The long-term potential benefit associated with GCTB is the greater nutrient availability, as it is speculated that burning results in either the volatilization of elements and/or their immobilization through the formation of basic oxides.

Organic carbon levels increased by 4 t/ha under the GCTB treatments compared with the burnt over a 9-year period. In contrast, under a grass ley system, organic carbon increased by 9 t/ha over a 6-year period compared with continuous cane, clearly demonstrating the superior nature of grass leys in sequestering carbon in this tropical environment. Charge fingerprints were derived from bulk samples collected from each of the trials. The divergence between the charge fingerprints due to treatment was greatest in the surface 10 cm, but this diminished with depth indicating that the influence of trash retention or grass leys on soil charge characteristics was confined to the surface layer. The increases in CEC at pH 5.5 of the GCTB treatment compared with the burnt, and of the grass ley treatment compared with the continuous cane were estimated to be 0.67 and 0.75 cmol_c/kg, respectively, over the 0-10 cm depth interval. Plots of ECEC versus CEC clearly indicated that 9-31% of the cations measured were not associated with the exchange complex and would, therefore, be subject to leaching loss. Whilst there was a higher net acidification rate under trash retention and grass ley production systems when compared to burnt or continuous cane systems, this must be viewed in the context of an overall enhancement in the surface charge characteristics of the soil through soil organic matter increase. Through a routine liming program instituted at plough out, acid generated over the rotation can be neutralized and, therefore, should not be viewed as a potential problem. In contrast, the generation of increased surface charge can only be viewed as beneficial. However, the increase in charge characteristics associated with elevated soil organic matter is transient. When the system is perturbed through cultivation, a significant proportion of this increased charge will be lost through organic matter mineralization.

6.0 TRASH DECOMPOSITION, SOIL SODICITY AND THE EFFICACY OF CALCIUM-BASED AMENDMENTS FOR AMELIORATING SODIC SOILS

6.1 Introduction

The suite of exchangeable cations has a marked influence on the physical properties of soil. Soils are termed 'sodic' if more than 6% of the cation exchange capacity is taken up by sodium. The higher the proportion of sodium, the more sodic the soil, and the worse its physical properties from a cropping point of view. Sugarcane yield reaches zero when surface soil sodicity or exchangeable sodium percentage (ESP) is greater than about 30% (Nelson and Ham 2000). Sodic soils cover a considerable proportion of all sugarcane

growing areas in Australia, except for the wet tropics (Nelson *et al.* 2001). In addition to causing crop management problems, cultivation of sodic soils can lead to excessive turbidity in runoff, with consequent detrimental impacts on downstream ecosystems.

Ca-containing ameliorants are used to reduce sodicity. The Ca replaces exchangeable Na, which must be leached from the soil. Trash is also known to improve crop performance on sodic soils, largely due to its ability to retain water and reduce evaporation losses.

The aim of this section was to examine interactions between Ca-containing ameliorants (lime and gypsum) and trash when applied to sodic soils. Interactions between Ca-containing ameliorants and molasses were also examined, as molasses has shown promise as an ameliorant, and was expected to interact with gypsum and lime. Soil organic matter has an important influence on the cation exchange properties of soil, and, therefore, influences the tendency of a soil to become or remain sodic (Nelson and Oades 1998). The effect of trash management on cation exchange properties of soil was examined in some detail. Sections of this work were carried out by Ahmad Suriadi in the University of Adelaide as part of his Masters-degree program.

6.2 Trash decomposition and soil sodicity

In Australia, significant areas of sodic soils (exchangeable sodium percentage, ESP, > 6) are used for cropping. Sugarcane is grown on soils with various degrees of sodicity, including those where sodicity is a serious limit to crop production (Nelson and Ham 2000; Nelson *et al.* 2001). The main reason for this limitation of crop growth is the adverse effect of sodicity on soil structure and the movement of water, air and roots through the soil. Management options aim at minimising these negative effects (Nelson *et al.* 2001).

Mechanical harvesting of unburnt sugarcane leaves large amounts of trash on the soil surface, e.g. 6-25 t dry matter/ha, most of which decomposes during the 12 months after harvest (Robertson 2003). Decomposition of sugarcane trash has not, however, been measured in sodic soils. It is important to know whether soil sodicity affects the rate of trash decomposition because there may be consequences for nutrient (particularly nitrogen) availability, and also the ease with which farming operations such as irrigation or cultivation can be performed. The effect of sodicity on decomposition also needs to be known to improve confidence in the use of simulation models, such as APSIM (Thorburn *et al.* 2001), to predict the effects of trash management on sodic soils.

In studies with other crop residues, sodicity has been shown to increase, decrease, or have no effect on decomposition (Nelson and Oades 1998). The objective of this study was to measure the effect of soil sodicity on decomposition of sugarcane trash.

6.2.1 Methods

Experimental details

Sugarcane (cultivar Q124) was grown in 40-cm diameter pots in a glasshouse for 1 year, fertilised with weekly applications of a mixed fertiliser including ¹⁵N-labelled ammonium sulfate. At maturity, the green and senesced leaves, which normally become trash during mechanical harvesting of the cane, were collected. They were dried in an oven at 65°C for

2 d, shredded through a cutter-grinder (2-10 cm long and 0.2-1.0 cm wide), mixed thoroughly, and subsampled. The fragment size of the subsample was then reduced to < 5 cm by cutting with scissors. Analysis of the trash gave the following results: total C, 36.3%; total N, 0.57%; C:N ratio, 64; water-soluble C, 3.9%; water-soluble N, 0.2%; and ¹⁵N atom excess, 6.8%.

Soil was collected from the 0-15 cm depth of cane fields at three locations in northern Queensland (Table 22). It was dried in an oven at 45°C for 6 d, then crushed to pass through a 2-mm screen, and sampled for analyses. The soils were then weighed into 13 cm-diameter plastic pots with holes on the bottom (1 kg /pot). The following treatments were then imposed on an equal number of pots of each soil: (1) Soil left untreated ('Original', T_O); (2) Soil made non-sodic, target ESP < 3 ('Non-sodic', T_{NS}); and (3) Soil made sodic, target ESP 20 ('Sodic', T_S).

Treatments T_{NS} and T_S were achieved through a three-stage leaching process: Stage 1 - the pots of soil were placed in a 30 L tub containing a 4 cm depth of the first solution (SAR 0, 0.5 mol_c/L calcium chloride for T_{NS} treatment; SAR 20, 0.5 mol_c/L calcium chloride and sodium chloride for the T_S treatment). More solution was gradually added to the tub over 1 d to bring it close to the height of the soil surface, and the pots left to soak for another day. The pots were then removed from the tub and one saturation volume of the solution slowly poured over the soil, whilst maintaining saturation. Stage 2 - the above procedure was repeated with the second solution (SAR 0, 0.1 mol_c/L calcium chloride for T_{NS}; SAR 20, 0.1 mol_c/L calcium chloride and sodium chloride for T_S). Stage 3 - Over 4 d, deionised water (equivalent of 8 soil saturation volumes) was poured on to the soil in each pot and allowed to drain. The pots were kept covered to minimise evaporation.

Table 22 Soils used in the experiment (0-15 cm depth)

	Burdekin	Mackay	Proserpine
Location	147.23 E, 19.50 S	149.07 E, 21.10 S	148.55 E, 29.39 S
Texture	sandy clay loam	sandy clay loam	loamy sand
Particle size (%sand, %silt, %clay)	53, 25, 22	56, 28, 16	82, 13, 5
Total N (%)	0.057	0.058	0.056
Total organic C (%)	0.68	1.08	0.71
C:N ratio	12	19	13
CEC: clay ratio	0.53	0.15	0.20

The pots were then weighed to estimate soil water holding capacity (WHC). The soils were again dried at 45°C for 6 d, crushed to < 2 mm, and thoroughly mixed within treatments. All treatment x soil type combinations were sampled for analyses.

Sugarcane trash (10 g) and soil (900 g) were mixed and added to 13-cm-diameter plastic pots with holes on the bottom, to provide a trash:soil ratio similar to field conditions (10 t dry matter /ha). The pots were shaken gently to settle the contents to a bulk density of 1.1 g/cm³. Water was applied to around 80% of field capacity. Enough pots were prepared for four replicates of each treatment x soil combination, plus four blanks (empty pots).

The pots were arranged in a glasshouse under shade cloth, in a randomised complete block design. Every 7-12 d, they were watered (usually to around 80% of saturation, but to >100% of saturation on several occasions) using filtered rainwater with electrical conductivity of 0.01 dS/m. The soils dried out considerably between waterings. Temperature was not controlled and varied between 17 and 43°C.

The T_S treatment was re-imposed on day 41 because the soil was found to be less sodic than expected. This time, one soil saturation volume of SAR 50, 0.5 mol/L sodium/calcium chloride solution was leached through the soil, followed by one soil saturation volume of SAR 50, 0.1 mol/L sodium/calcium chloride solution, then 2-3 saturation volumes of deionised water. Drainage ceased in the Burdekin soils, and became very slow in the Mackay soils, soon after the second volume of water was applied. Several days later, small samples of soil (5-7 g) were taken from each of the T_S pots (replicates were pooled) for measurement of exchangeable cations.

Carbon dioxide (CO₂) being released from the pots was trapped on four occasions (days 3, 20, 56 and 77) by placing vials containing 20 mL of 1M sodium hydroxide on the soil surface and sealing the pots in heavy-duty zip-seal plastic bags for 2-3 d.

At the end of the experiment (day 136), the contents of the pots were cut in half vertically: one half was retained for soil analyses, and the other half was washed under running water and the remaining trash caught on a 2-mm sieve.

Analytical methods

Carbon dioxide in the sodium hydroxide traps was determined by titration against dilute hydrochloric acid (Zibilske 1994). Total C and N analyses were done on dried subsamples of soil ground to < 250 µm and trash ground to < 500 µm. Total N and C in soil and trash were determined using a Leco CNS 2000 combustion analyser. Soil inorganic N was determined by extracting 15-g portions of moist soil in 2M potassium chloride, and measuring ammonium and nitrate in the filtered extracts by automated colorimetric analysis (Rayment and Higginson 1992, method 7C2). Soil pH and electrical conductivity (EC) were measured in 1:5 soil-water extracts (Rayment and Higginson 1992, method 3A1). Cation exchange capacity (CEC) and exchangeable calcium, magnesium, potassium and sodium (Ca_E, Mg_E, K_E, and Na_E) were determined by compulsive exchange with silver-thiourea (Rayment and Higginson 1992, method 15F1). Clay content was measured by the pipette method. Spontaneously dispersible clay was measured using turbidimetry according to the method of Rengasamy *et al.* (1984). Calibration curves of turbidity vs. clay concentration were prepared for each soil.

Calculations and statistical analyses

Cumulative CO₂ evolved during the experiment was estimated from the area under the curve of CO₂ accumulation.

The main effects and interactions of leaching treatment and soil type were tested for each sampling time separately by two-way analysis of variance for a randomised complete block design using the Systat[®] program (SPSS Inc.). Relationships between selected variables were investigated using Pearson's correlation. Unless stated otherwise, effects were taken as significant where Bonferroni probabilities were ≤ 0.05.

Table 23 Soil properties before trash addition (day 0), after sodic treatments re-imposed (day 41), and at the end (day 136) of the experiment. Subscript _E denotes exchangeable cations. T_O, T_S, T_{NS} denote unamended, sodic, and non-sodic treatments, respectively. Least significant differences (LSD) are for soil x treatment interaction

Treatment	Burdekin			Mackay			Proserpine			LSD _(0.05)
	T _O	T _S	T _{NS}	T _O	T _S	T _{NS}	T _O	T _S	T _{NS}	
Day 0										
CEC (cmol(c)/kg)	11.7	13.2	11.7	2.5	3.6	4.1	1.1	1.5	1.7	
Ca _E (cmol(c)/kg)	5.6	11.0	11.0	1.4	3.3	3.8	0.8	1.4	1.7	
Mg _E (cmol(c)/kg)	4.5	0.6	0.3	0.7	0.0	0.1	0.2	0.0	0.0	
K _E (cmol(c)/kg)	0.2	0.2	0.2	0.2	0.1	0.1	0.1	0.0	0.0	
Na _E (cmol(c)/kg)	1.4	1.4	0.2	0.2	0.2	0.1	0.0	0.1	0.0	
ESP	12.0	10.6	1.7	8.0	5.5	2.4	0.0	6.7	0.0	
pH	6.51	7.12	6.68	4.45	5.07	4.83	4.35	5.61	5.24	
EC (dS/m)	0.18	0.13	0.09	0.04	0.04	0.03	0.20	0.03	0.03	
Soil inorganic N (µg/g)	38.0	6.9	5.5	13.7	9.4	7.5	99.0	4.4	3.1	
Saturated WHC (% w/w)	43	39	40	38	33	35	26	26	26	
Total pore volume (% v/v)	66	62	63	61	56	58	48	48	48	
Day 41										
CEC (cmol(c)/kg)		20			5			3		
Ca _E (cmol(c)/kg)		13.2			3.7			1.7		
Mg _E (cmol(c)/kg)		0.9			0.2			0.2		
K _E (cmol(c)/kg)		0.5			0.3			0.3		
Na _E (cmol(c)/kg)		5.3			1.0			0.5		
ESP		26.6			19.2			18.5		
Day 136										
CEC (cmol(c)/kg)	15.5	17.5	15.4	9.6	10.0	10.2	7.7	8.3	7.2	2.9
Ca _E (cmol(c)/kg)	7.2	11.9	11.2	2.0	3.4	3.4	1.4	1.6	1.7	0.4
Mg _E (cmol(c)/kg)	4.1	0.8	0.7	0.6	0.1	0.2	0.2	0.1	0.2	0.04
K _E (cmol(c)/kg)	0.4	0.4	0.4	0.2	0.2	0.2	0.1	0.1	0.1	NS
Na _E (cmol(c)/kg)	0.5	1.3	0.2	0.2	0.6	0.2	0.1	0.3	0.1	0.22
ESP	3.2	7.4	1.3	2.1	6.0	2.0	1.3	3.6	1.4	
pH	6.43	7.05	6.51	5.00	5.34	5.20	5.47	5.76	5.67	0.18
EC (dS/m)	0.09	0.11	0.08	0.02	0.06	0.03	0.03	0.06	0.04	0.03
Soil inorganic N (µg/g)	3.5	1.3	0.9	1.8	1.5	1.6	1.0	1.3	1.6	1.4
Clay (%)	21.6	26.3	16.8	14.8	14.6	14.4	4.7	4.6	4.7	5.5
Dispersible clay (% total)	0.44	0.96	0.23	1.52	1.32	1.34	1.34	1.92	1.65	1.3
Trash retrieved (%)	48	57	37	46	50	35	64	70	59	NS
Recovery of ¹⁵ N (%)	75	85	101	83	107	75	81	74	64	NS

6.2.2 Results

Soil type effects

In the original soils, clay content, CEC, ESP, and WHC decreased in the order Burdekin > Mackay > Proserpine (T_O, Table 23). Soil pH was greater in the Burdekin soil than in the Mackay and Proserpine soils. The Burdekin soil also had a higher ratio of CEC to clay content than the Mackay and Proserpine soils (Table 22). The values of the ratio indicated mixed mineralogy in the Burdekin soil and kaolinite-dominated mineralogy in the other two (Shaw *et al.* 1998). Total soil N was similar in the three soils, but total organic C and C:N ratio were greater in the Mackay soil (Table 22). Inorganic N concentration in the Proserpine soil was more than twice that in the Burdekin and Mackay soils, reflecting

recent fertiliser application. Inorganic N in all soils was dominated by nitrate (data not shown).

Leaching treatment effects

The day 0 soil analysis showed that T_{NS} had effectively reduced the original soil sodicity levels, rendering the T_{NS} soils non-sodic (ESP 0-2) (Figure 10). At day 0, the T_S treatment made the Proserpine soil mildly sodic, but had little effect on sodicity in the Burdekin and Mackay soils. Re-imposition of T_S on day 41 markedly increased soil ESP, making all the T_S soils highly sodic (ESP 18-27) (Figure 10). Following both leaching treatments, the Burdekin soil attained a higher ESP than the Mackay or Proserpine soils for the same level of solution SAR, indicating a higher Gapon coefficient for the Burdekin soil (Table 24). The Gapon coefficients were higher following the second leaching due to the higher concentration of the solution used. The original soils were of intermediate sodicity (ESP 0-12).

Soil pH was increased by both T_S and T_{NS} in the Burdekin and Mackay soils, and by T_S only in the Proserpine soil (Table 23).

Soil inorganic N was reduced by the leaching treatments, due to loss of nitrate. Total soil N and organic C were not significantly affected by the leaching treatments (data not shown).

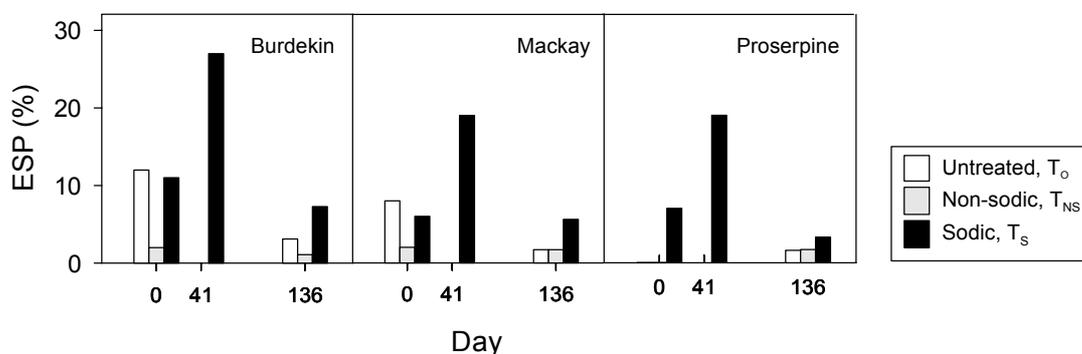


Figure 10 Exchangeable sodium percentage (ESP) before trash addition (day 0), after sodic treatments re-imposed (day 41), and at the end of the experiment (day 136)

Table 24 Gapon coefficients for the T_S soils following the first (day 0) and second (day 41) leaching

Time	Burdekin	Mackay	Proserpine
Day 0 (SAR 20)	0.0064	0.0031	0.0036
Day 41 (SAR 50)	0.0080	0.0054	0.0059

Trash decomposition

In the T_O treatments, CO_2 evolution was greater in the Burdekin soil than the Mackay and Proserpine soils. In the Burdekin soil, CO_2 evolution was reduced in the T_S relative to T_{NS} and T_O treatments. The leaching treatments had no significant effect on CO_2 evolution from the Mackay and Proserpine soils (Figure 11).

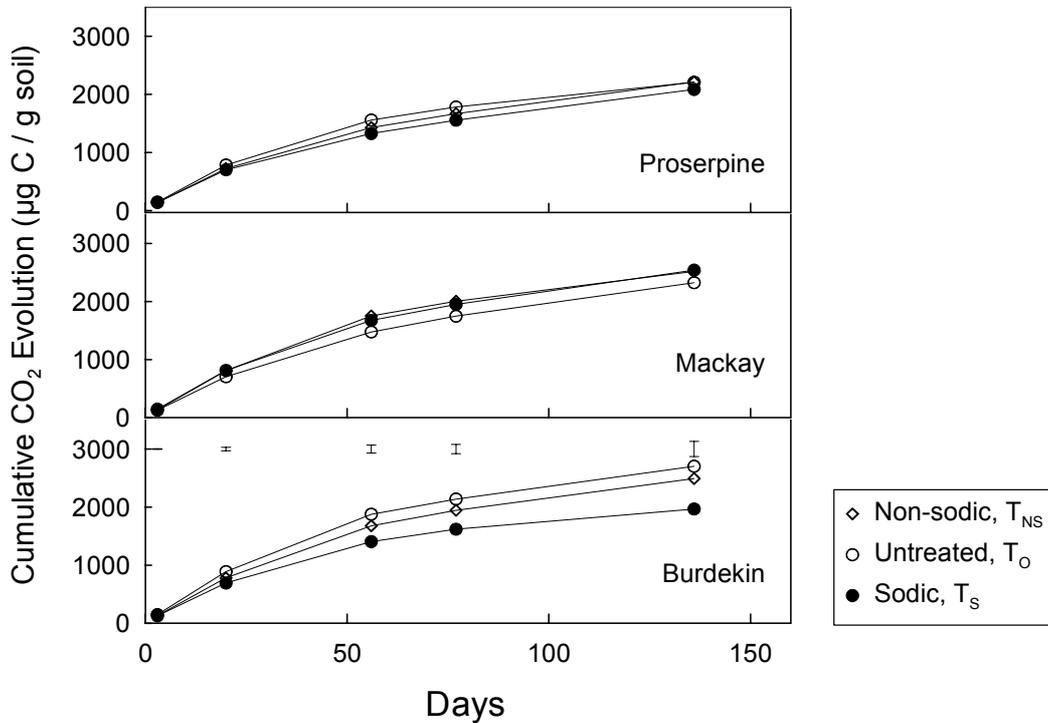


Figure 11 Cumulative CO_2 evolution as affected by soil and treatment. Bars on Burdekin graph are LSD ($P=0.05$) for treatment x soil interaction. Treatments were not significantly different in Proserpine and Mackay soils

The trash retrieved from the soil at the end of the experiment was very variable among replicates, but tended to be greatest in the T_S and least in the T_{NS} treatment, though differences were not statistically significant (Table 23). Average dry matter recovery was 52%. Recovery of trash- ^{15}N at the end (mean 85%) was similarly variable among replicates and did not differ significantly among treatments.

Trash retrieved was weakly negatively correlated with cumulative CO_2 released during the experiment ($r = -0.43$, $P < 0.05$ for all data; $r = -0.62$, $P < 0.07$ for replicate means). Neither CO_2 evolved nor trash dry matter retrieved were related to trash ^{15}N retrieved.

Treatment effects on day 136

The treatment effects on day 136 generally reflected those on day 0 (Table 23). Total N and organic C showed no significant change in time (data not shown). Soil pH increased slightly in the Mackay and Proserpine soils, but was unchanged in the Burdekin soil

(Table 23). The Burdekin and Proserpine soils originally had relatively high EC, and this was significantly reduced by day 136 ($P < 0.05$).

Soil CEC was universally greater on day 136 than on day 0 ($P < 0.05$). Soil CEC was correlated with clay content ($r = 0.91$, $P < 0.05$).

In all soils, ESP in the T_S treatment was reduced to < 8 on day 136 ($P < 0.05$). The ESP in T_{NS} and T_O soils generally declined with time, except in the Proserpine soil (Figure 10). The percentage of this clay that was spontaneously dispersible was $< 2\%$ in all cases, and decreased in the order: Proserpine $>$ Mackay $>$ Burdekin. Spontaneously dispersible clay was not significantly affected by the treatments.

The T_S treatment increased total clay content in the Burdekin soil ($P < 0.05$), but not in the Mackay and Proserpine soils. The effect in the Burdekin soil probably occurred because the standard dispersion treatment for measuring particle size distribution was not adequate to fully disperse the soil in its non-sodic state.

In the Burdekin soil only, final soil Na_E , ESP, clay content and dispersible clay were negatively related with final cumulative CO_2 released ($r = -0.74$, -0.75 , -0.72 , -0.80 , respectively, $P < 0.05$).

By day 136, soil inorganic N was reduced to a very low level in all pots (Table 23).

Soil-water content measured at the end of the experiment and estimated by weighing the pots during the experiment followed the same trends as WHC. Generally, Burdekin $>$ Mackay $>$ Proserpine, and $T_{NS} > T_S$ (effects did not always attain significance at $P < 0.05$, but trends were consistent, data not shown). Final soil water content was correlated with clay content ($r = 0.89$, $P < 0.05$).

6.2.3 Discussion

The experimental approach used in this study allowed the effects of soil sodicity on trash decomposition to be measured without the confounding influences of soil, trash, and environment factors, which would have been unavoidable in a field study.

The T_S treatment decreased trash decomposition in the Burdekin soil by about 18% (*cf.* CO_2 release from T_{NS}). The effect was related to high Na_E and high ESP in the T_S soil. In contrast, T_S had no effect on decomposition in the Mackay and Proserpine soils. This may have been partly due to lower levels of sodicity being achieved in these soils, and partly due to other soil factors. For example, the amount and type of clay has been shown to influence the effects of sodicity (Churchman *et al.* 1993; Nelson *et al.* 1997). The Burdekin soil, which had the greatest clay content and the highest pH, sealed up when the T_S was imposed, and thereafter water penetration was very much slower than in the Mackay and Proserpine soils (data not shown). This response in the Burdekin soil can be attributed to blocking of the soil pores because of dispersion and swelling of the clay at high ESP and pH (Churchman *et al.* 1993). Thus, it is likely that much of the sodicity effect on decomposition was due to impaired aeration.

The T_S treatment was strongly sodic according to ESP (18-27). Milder sodicity (ESP 8-12) had no effect on trash decomposition (CO₂ production) in any of the soils, as can be seen by comparing the T_O and T_{NS} treatments (Figure 8). This experiment was watered using rainwater of low electrical conductivity, therefore the effect of high ESP on soil hydraulic properties would have been maximised (Quirk and Schofield 1955). The effects of soil sodicity on decomposition would probably be even less in the field than in this study, because trash is usually left on the surface rather than being mixed with the soil.

Effects of sodicity on organic matter decomposition and soil microbial populations have been little studied. However, Nelson *et al.* (1996, 1997) found that the size of the soil microbial biomass was unaffected, and decomposition of added legume material was either unaffected, slightly increased, or slightly decreased by increasing soil sodicity.

The maximum ESP achieved by T_S in the Burdekin soil (27) was very high. Under field conditions, such a soil would present serious difficulties for crop production (Nelson and Ham 2000). An 18% reduction in trash decomposition, as found in this study, may therefore be of relatively little consequence to the farmer.

The T_S treatment resulted in a higher ESP in the Burdekin soil than the other soils because of its higher Gapon coefficient (Table 24). The difference in coefficients may have been partly due to differences in mineralogy, but also to the different ratios of clay to organic matter. Cation exchange sites on organic matter tend to have a lower affinity for Na than exchange sites on clay minerals (Nelson and Oades 1998). Recent work has shown that retention of sugarcane trash in long-term field trials reduces the tendency of soil to become sodic (Suriadi *et al.* unpubl. data).

Trash retrieved at the end of the experiment was very variable among replicates because the pot contents were divided without mixing, and because of the inconsistent ease of trash removal from the different soil types. This resulted in the poor correlation between trash retrieved and cumulative CO₂ evolution. We recommend using a larger trash fragment size than used in this study when decomposition is to be measured by similar retrieval techniques.

Mineralisation of native soil C can be affected by sodicity (Nelson *et al.* 1996) and may have differed among soils and leaching treatments in this experiment. However, any significant effect on the experimental results would have been unlikely, because mineralisation of soil C is much lower than the mineralisation of C in trash (see Section 8).

The large decrease in soil inorganic N during the experiment probably resulted from a combination of microbial immobilisation during decomposition of the trash, leaching (Proserpine soil), and denitrification (Mackay and Burdekin). In related studies, sugarcane trash was shown to promote microbial immobilisation of N, but applications of up to 400 kg N/ha as urea did not increase the rate of trash decomposition (Fiona Robertson, unpubl. data). Thus, the large concentration of inorganic N in the T_O Proserpine soil is not likely to have influenced trash decomposition in this experiment.

Decomposition of the trash changed the cation exchange properties of the soils in several ways. The increase in CEC from day 0 to day 136 must have been due to the production

of organic materials with negative charge. In the Mackay and Proserpine soils there may also have been a contribution from the effect of increased pH on variable charge components of the CEC. Increases in Ca_E , especially in the Burdekin soil, must also have originated from the trash. The trash contained approximately 0.28% Ca, which, at the trash addition rate used, corresponds to a Ca addition of 1.5 cmol_c/kg soil. The increase in Ca_E occurred within the first 40 d. The amount of K added in trash was 0.74 cmol_c/kg soil. Increases in soil contents of exchangeable cations were greatest in the Burdekin soil, probably due to less leaching loss than in the other two soils. Substantial decreases in ESP of the Burdekin soil from day 0 to day 41 were due mostly to the increase in Ca_E rather than loss of Na.

6.3 Effects of lime, gypsum, trash and their interactions on sodic soils and sugarcane growth

The aim of this investigation was to determine whether interactions between lime, gypsum and trash influence the amelioration of alkaline and acid sodic soils.

6.3.1 Methods and material

The trial was set up as a twin factorial design, with two soils (Burdekin or Proserpine), and nine treatments involving ameliorant (nil, gypsum or lime) and trash (nil, surface or incorporated). There were four replicates, resulting in a total of 72 pots. Each pot was planted with a one-eye sett of sugarcane (Q117) on 5 July 1999. The pots were placed on Besser blocks on a gravel bed at a rectangular spacing of 1 x 0.7 m, and were surrounded by a row of guard pots. Wire fences were erected to prevent the pots from falling over. Many did fall over during Cyclone Tessi, but virtually no soil was lost and the pots were subsequently righted. The pots were made from PVC pipe 0.3 m in diameter and 0.6 m long, with a perforated base. Subsoil and then topsoil from each of the two soils was placed in the pots in layers, each 0.25 m deep. Three doughnut-shaped plastic baffles were taped to the inside of the pot at evenly spaced depths as the soil was placed in the pots. These were to prevent roots moving down the side of the pots. By the end of the experiment the soil had slumped by an average of 80 mm.

The gypsum (Winton-mined gypsum, with 13.5% S and 17.4% Ca) application rate was equivalent to 10 t/ha (71 g/pot). The lime (Zinaback lime, 37% Ca) was added at an equivalent rate of Ca to the gypsum (4.7 t/ha, 33.4 g/pot). Trash was chopped into pieces 50-70 mm long and added at 10 t/ha (DM). In the incorporated treatments it was incorporated into the top 100 mm. Urea was added initially at 478 kg/ha (3.4 g/pot) and a slow release fertilizer was added in the second year. The pots were irrigated daily or twice daily using drippers and were hand weeded from time to time. The pots were initially set up at BSES Burdekin, but were shifted to CSIRO Davies lab on 26 October 1999.

Cane length was measured on 7 February 2000. It varied from 0-700 mm, with no significant effect of treatments. On 14 February 2000, following a rainfall event, leachate was collected. The first harvest was conducted on 2 October 2000. Trash was removed and the green leaves plus cabbage and stalks were weighed separately. The green leaves plus cabbage was mulched and occasional samples taken for water content measurement.

The stalks were analysed for CCS using a small mill and assuming constant fibre content. Trash was added again to appropriate pots, to the surface in both ‘surface’ and ‘incorporated’ treatments. The second harvest was conducted on 18 April 2001, in the same way as the first. The soil was then sampled, taking 3 x 70 mm cores per plot and bulking them for four depths.

6.3.2 Results and discussion

The treatments had little or no significant effect on cane yield or CCS during the experiment, due largely to high variability (Figures 12 and 13). The high variability appeared to be due to cracks forming in some pots; in those pots irrigation water and ameliorants flowed preferentially down the cracks. Growing sugarcane in sodic soils these large pots was extremely difficult to manage. Leachate properties reflected soil type but not treatment (Figure 14). The final soil sampling indicated effects of gypsum and lime on the Burdekin soil, but no interactions between trash treatment and gypsum or lime. In the Burdekin soil, lime generally decreased ESP at the surface and gypsum generally decreased ESP at depth (Figure 15). Leaching alone had considerably reduced ESP in all treatments.

The experiment did not result in clear results, because of the difficulty in maintaining uniform water content and flow through the soils. Subsequent pot trials were carried out in smaller pots without plants.

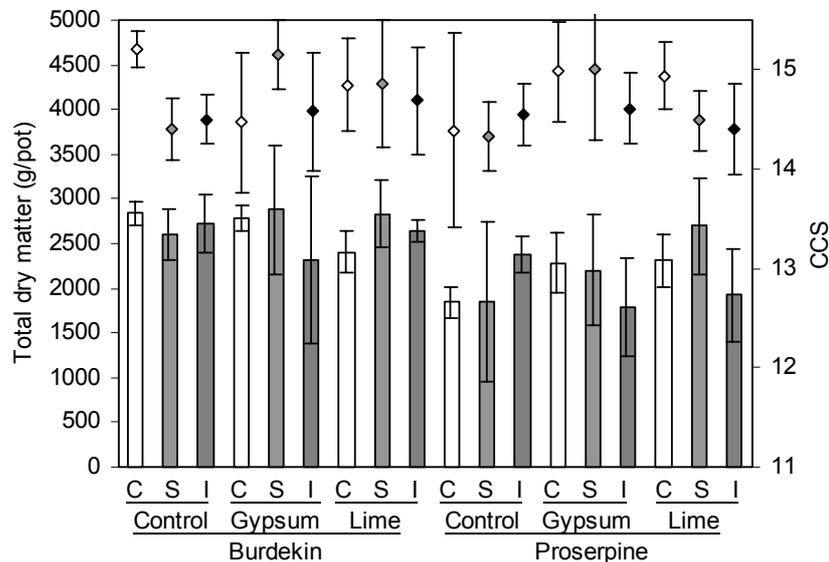


Figure 12 Total above-ground dry matter (bars) and CCS (points) of cane at first harvest (October 2000). Trash treatments are indicated by C (control), S (surface-applied) and I (incorporated). Error bars show standard deviations

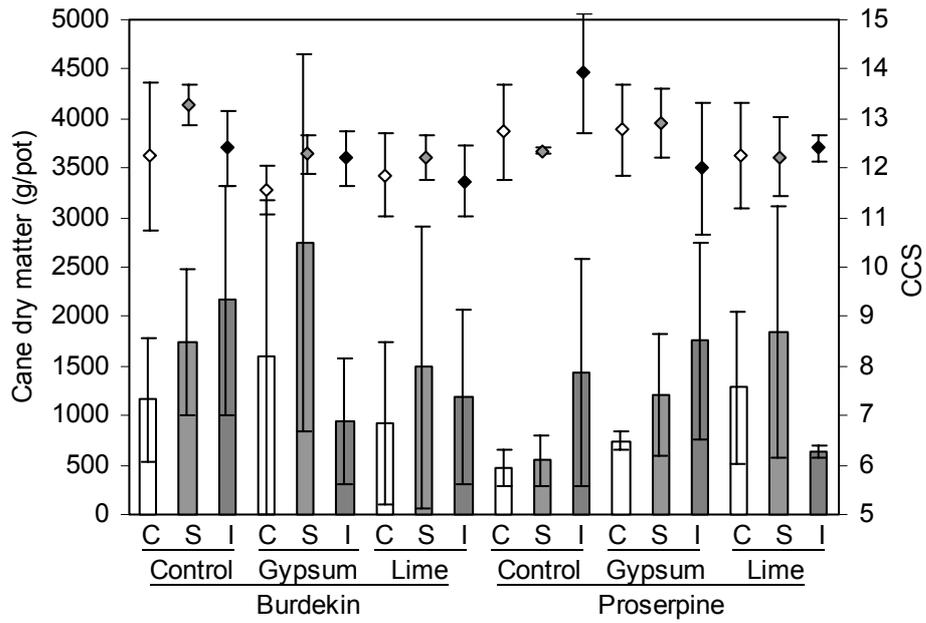


Figure 13 Cane dry matter (bars) and CCS (points) at the second harvest (April 2001). Trash treatments are indicated by C (control), S (surface-applied) and I (incorporated). Error bars show standard deviations

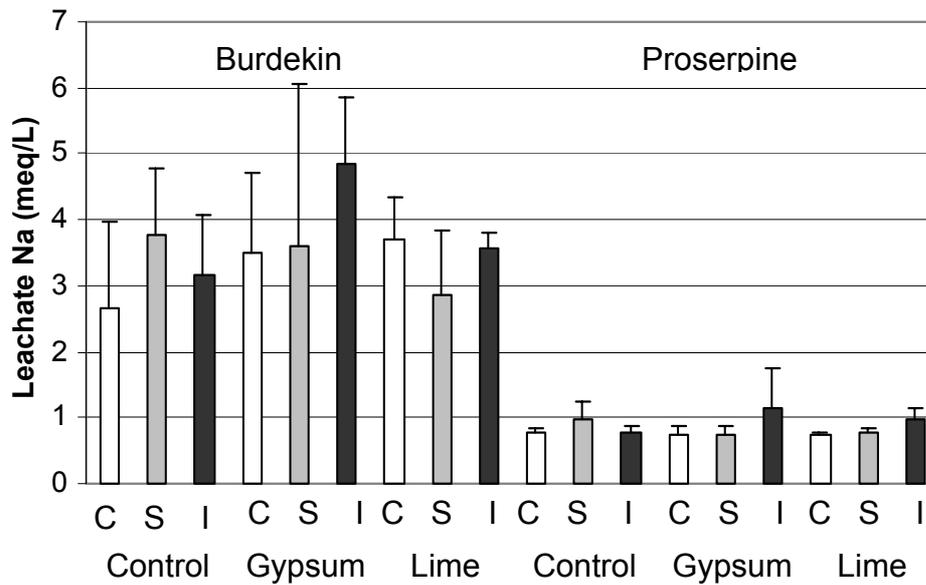


Figure 14 Sodium concentration of pot leachate collected on 15 February 2000. Letters indicate the trash treatments: C, control; S, surface applied; I, incorporated

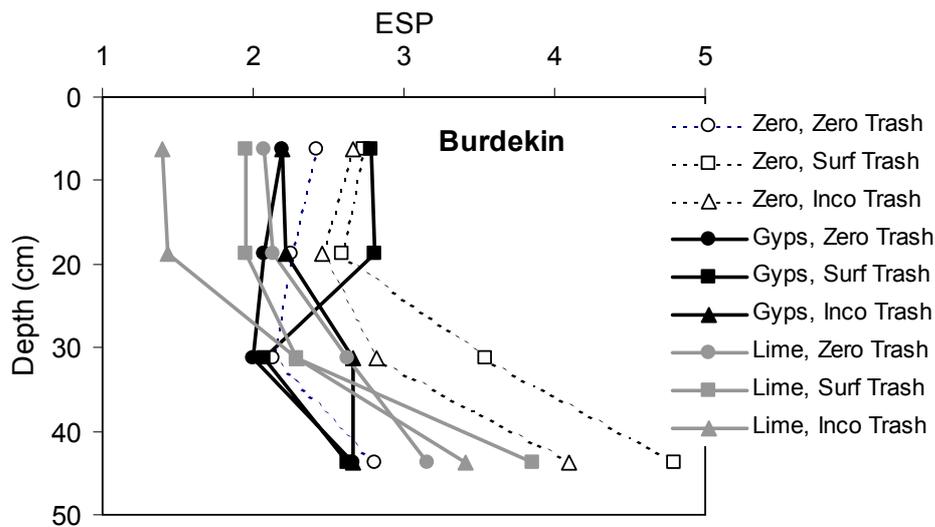


Figure 15 Exchangeable sodium percentage (ESP) in the Burdekin soil at the end of the experiment

6.4 Structural stability of sodic soils as influenced by gypsum/molasses

Stable structure and structural stability are essential for a fertile soil. One of the measures of stable structure is stability of soil aggregates in water. Such changes in water-stable aggregation have generally been correlated with the quality and quantity of organic matter in the soil (Tisdall and Oades 1982; Piccolo 1996; Puget *et al.* 2000).

Gypsum ($\text{CaSO}_4 \cdot 2\text{H}_2\text{O}$) and organic materials are commonly used as ameliorants to overcome sodicity and related problems. In the sugarcane-growing districts of Queensland, gypsum is available at about $\$90 \text{ t}^{-1}$, and a substantial source of organic material is available as molasses at about $\$45 \text{ t}^{-1}$. Molasses, a by-product of sugar production, contains carbohydrates (62%), protein (3-4%) and ash (8.6%) (Weber and Van Rooyen 1971), as well as cations (calcium, 0.07%; magnesium, 0.08%; potassium, 0.5%; sodium, 0.03%), nitrogen (0.39%) and water (20% w/w), and has a pH of about 5.5 (data from this work).

The aim of this work was to determine whether molasses alone or combined with gypsum, could improve the structural stability of two sodic soils used for sugarcane production.

6.4.1 Materials and methods

Soils

The soils used were the top layer (0-15 cm depth) of sodic soils from Proserpine and the Burdekin. The Burdekin soil is a mesonatric brown sodosol and the Proserpine soil a natric brown kurosol (Isbell 1996). Table 25 shows some of the physical and chemical properties of the soils. Both soils contained a range of clay minerals; the Burdekin soil is dominated by illite and illite-smectite, and the Proserpine soil by illite and kaolinite.

Table 25 Selected properties of soil used in this study

Soil	pH ^A	EC ^A (dS/m)	Cations (cmol(+)/kg)				CEC (cmol(+)/kg)
			Ca	Mg	K	Na	
Burdekin	8.0	0.2	6.5	3.9	0.5	0.9	10.9
Proserpine	4.9	0.1	0.9	0.4	0.4	0.4	2.3
	ESP	Org C (%)	Particle size (%)			Texture	
			Clay	Silt	Sand		
Burdekin	7.9	0.8	21.7	25.4	52.8	Sandy clay loam	
Proserpine	18.8	0.7	5.0	12.8	82.2	Loamy sand	

^A1:5 soil:water extracts.

Experimental design and treatments

The experiment had a factorial design with three factors (gypsum, molasses, and soil), each at two levels, and four replicates. Each experimental unit was a pot of soil, kept at 25°C for a period of 12 weeks. Data were analysed using the Genstat V Statistical Package (Genstat 5 Committee 1987).

The gypsum and molasses factors consisted of an application of 0 or an application equivalent of 10 t ha⁻¹ on an oven-dry basis (6.08 g pot⁻¹). For the molasses treatments, molasses was dissolved in water and sprayed uniformly onto a thin layer of soil, which was then dried and mixed before being placed in pots.

Pots (8.8 cm inner diameter, 10 cm long) contained 500 g treated soil, which produced bulk densities in the range of 1.4-1.5 g/mL. Fine mesh was placed over a funnel at the base of each pot to retain soil but to allow leachate to pass. All the pots were wetted to 85% of field capacity (FC) and maintained in this state for 7 d to provide time for dissolution, diffusion and microbial activity. Pots were leached with 0.5 pore-volumes of distilled water (88 mL or equal to 14.5 mm rainfall for the Burdekin soil and 72 mL or equal to 11.8 mm rainfall for the Proserpine soil). This water was applied as a spray over a period of 2.5-3 h. Leachate was collected and analysed for dissolved organic carbon (DOC), cations (Ca²⁺, K⁺, Mg²⁺ and Na⁺), pH and EC. The sodium adsorption ratio (SAR) of solutions was calculated from: $SAR = [Na^+]/([Ca^{2+}] + [Mg^{2+}])^{0.5}$, where cation concentrations are expressed in mmol/L. Between leaching events, pots were allowed to dry for 7-10 d, which was long enough to bring them to their original air-dry water contents. At this point, they were again leached with water, during which leachate was collected for analysis. These wetting and drying cycles were conducted to simulate the changes in soil solution concentration that occur in the field and to promote the migration of solutes and colloids that may impact on soil structural stability. Leaching events were conducted a total of five times and all pots were then air-dried for soil analysis after the last leachate had been collected.

Soil analyses

The pH and EC were determined using 1:5 soil:water extracts. Exchangeable cations and CEC were determined using methods described by Rayment and Higginson (1992). Exchangeable cations (Na⁺, K⁺, Ca²⁺, and Mg²⁺) were extracted with 1 M NH₄Cl at pH 7 and concentrations were measured using Inductively Coupled Plasma Atomic Emission Spectrometry (ICPAES). The CEC of the leached samples was then measured by

displacement of NH_4^+ with K-Ca displacing solution (15% KNO_3 and 6% $\text{Ca}(\text{NO}_3)_2 \cdot 4\text{H}_2\text{O}$) and subsequent measurement of NH_4^+ by auto-analyser. The organic carbon content was measured in a Leco furnace and the ESP was calculated from exchangeable sodium concentrations and the measured CEC.

To measure wet aggregate stability, 25 g of soil aggregates (5-10 mm) were wet-sieved for 10 min on a set of two sieves (0.25, 0.125 mm), following the method described by Kemper and Rosenau (1986). The fraction remaining on each sieve was collected, oven dried at 105°C and weighed. The fraction <0.125 mm was calculated by difference.

To measure spontaneous dispersion, 20 g of sieved soil (<2 mm) was placed in a 150 mL vial and 100 mL of deionised water was poured gently down the side of a vial, which was then allowed to stand for 24 h. The vial was then slowly inverted once and returned upright, giving just enough agitation so that the aggregates simply slid down the side of the vial. The supernatant was then stirred slowly and gently to allow the suspended matter to mix without further disturbance to the soil sediment in the container. After allowing an appropriate sedimentation time for particles >2 μm , the supernatant containing dispersed clay was drawn off and its concentration measured by turbidimetry. An individual calibration curve for both soils was used to convert turbidity to clay concentration. A similar procedure was used to measure mechanical dispersion. The 1:5 soil:water suspension was agitated on an end-over-end shaker for one hour at 60 rpm. The top 2 cm of suspension containing <2 μm material was drawn off after allowing an appropriate sedimentation time and measured by turbidimetry.

6.4.2 Results

Leachate properties

Figures 16 and 17 show the effects of molasses and gypsum on the properties of successive leachates from the Burdekin and Proserpine soils, respectively. The first leaching was conducted on soil that had been slowly wetted to 85% of field capacity; subsequent leachings were conducted on air-dried soil. After five leachings, there were significant changes in leachate properties imposed by molasses and gypsum treatments for both soils. As the fourth leachate was only analysed for EC and pH, data for this leaching are not presented. However, as the pH and EC values in the fourth leachate were intermediate between those of leachates 3 and 5, it is probable that SAR and DOC values were also intermediate between those of leachates 3 and 5.

In the Burdekin soil (Figure 16), pH and SAR were generally reduced when molasses and gypsum were added. However, in the case of the lighter-textured, more acid Proserpine soil (Figure 17), molasses tended to increase pH and SAR, whereas gypsum did the opposite. Molasses and gypsum increased the EC of leachate in both soils. For the Burdekin leachate, the EC was generally greatest in the gypsum-only treatment compared with either molasses-only or molasses+gypsum. The highest EC was found in the second leachate of the gypsum-only treatment and the lowest EC was found in the fifth leachate of the control. For the Proserpine soil, the EC was greatest for the molasses+gypsum treatment. The lowest EC was found in the first leachate of the control and the highest EC was found in the third leachate of the molasses+gypsum treatment. The average EC of Burdekin leachates ranged between 0.5 and 6.5 dS/m for all treatments and for the Proserpine soil, between 0.3 and 5.5 dS/m.

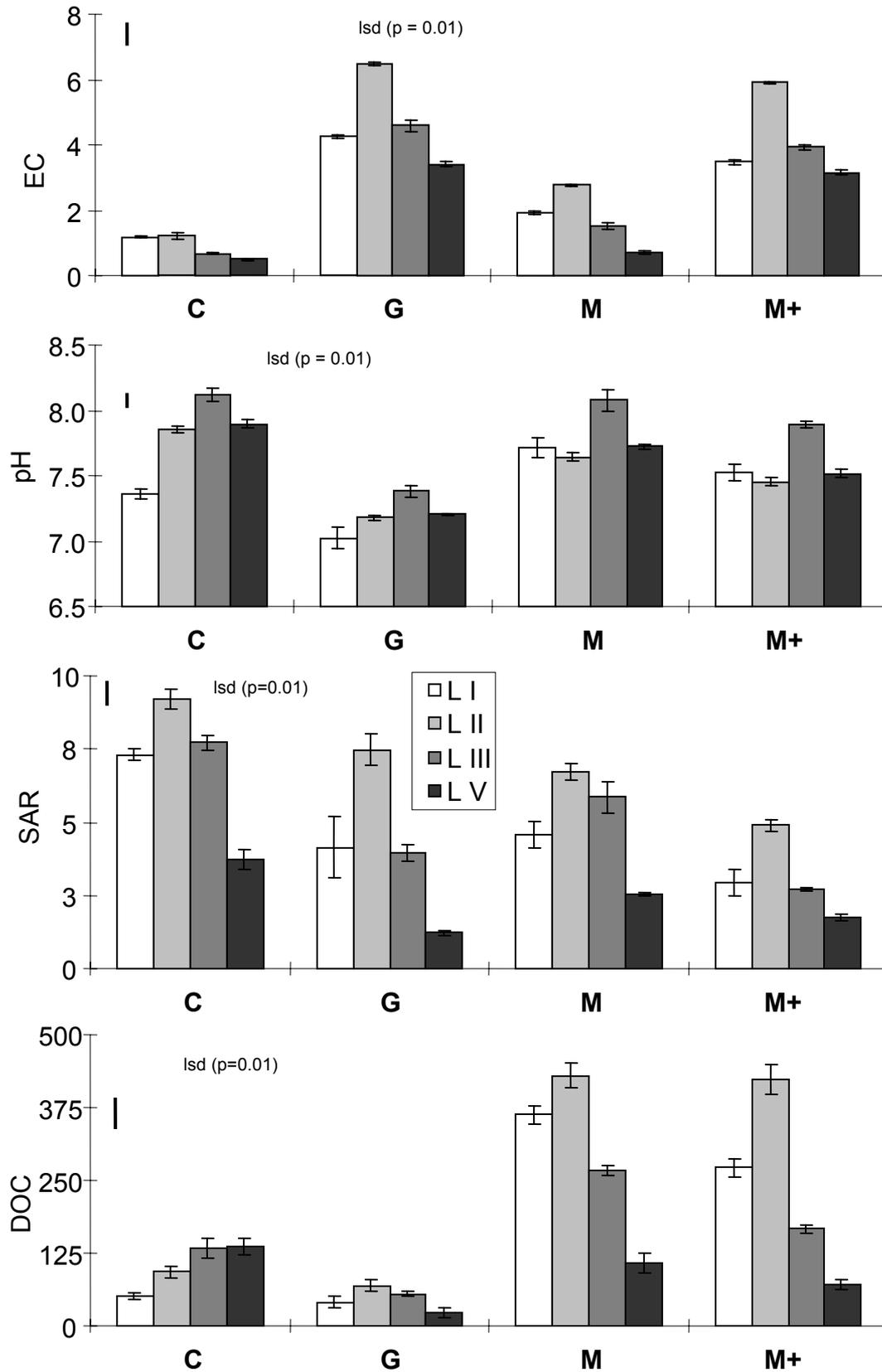


Figure 16 pH, EC, SAR, DOC in successive leachates of the Burdekin soil after molasses and gypsum treatments. C, M, G, M+G refer to control, molasses, gypsum and molasses+gypsum, respectively. L I-V are leachates from leaching events I-V

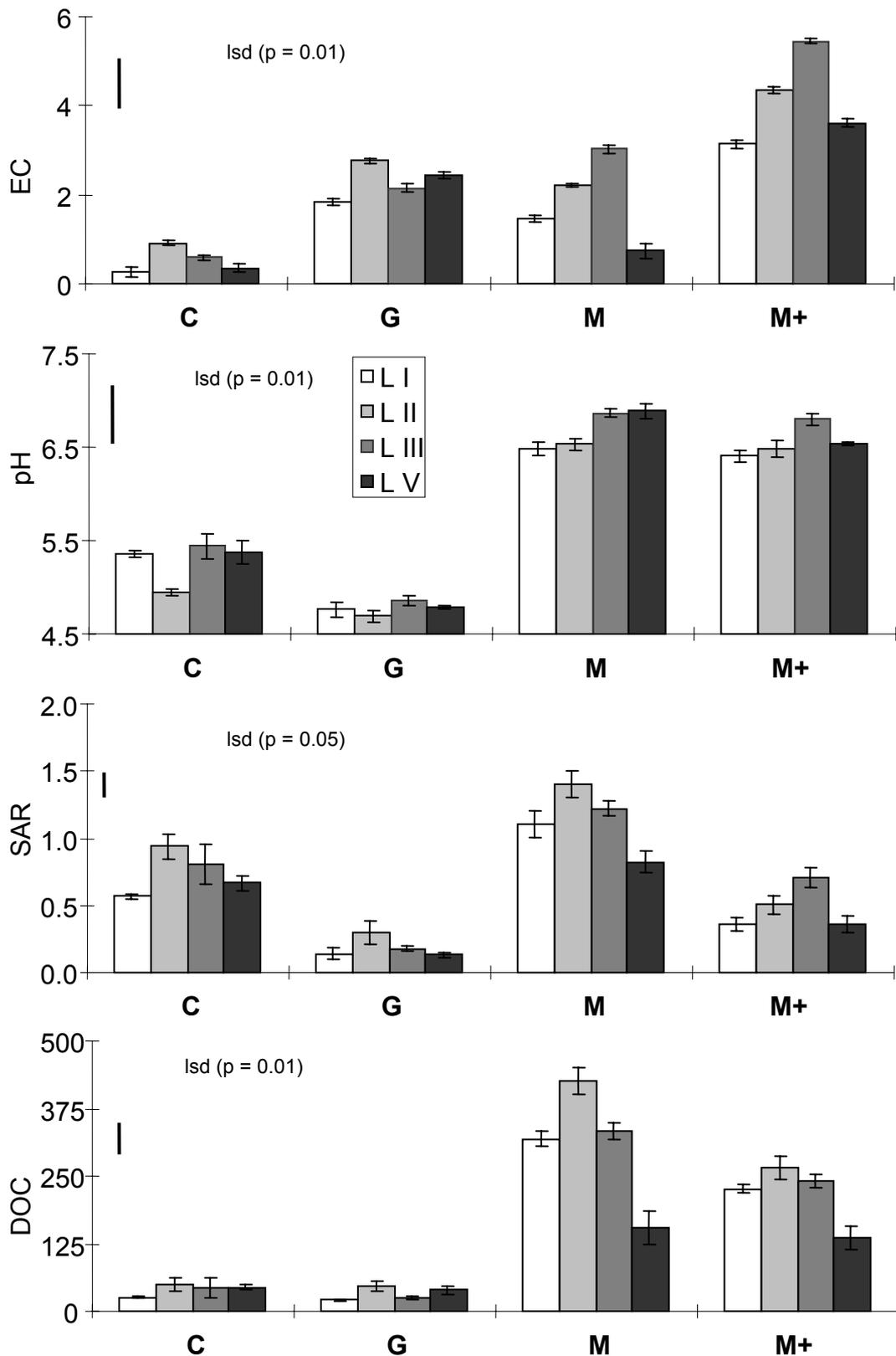


Figure 17 pH, EC, SAR, DOC in successive leachates of the Proserpine soil after molasses and gypsum treatments. C, M, G, M+G refer to control, molasses, gypsum and molasses+gypsum respectively. L I-V are leachates from leaching events I-V

Dissolved organic carbon (DOC) concentrations were high in molasses treatments. For the Burdekin soil, the highest DOC was found in the second leachate of the molasses-only treatment and the lowest DOC was found in the fifth leachate of the gypsum-only treatment. DOC generally decreased as leaching proceeded, except in the control. For the Proserpine soil, the lowest DOC was found in the first leachate of the gypsum-only treatment and the highest was found in the second leachate of the molasses-only treatment. In general, DOC was less in molasses+gypsum than in molasses-only treatments in both soils as calcium presumably immobilised some organic carbon.

Clay dispersibility

Both gypsum and molasses reduced spontaneous and mechanical dispersion substantially in both soils (Figure 18 and 19). In the Burdekin soil, the reduction in spontaneously dispersed clay compared to the control was dramatic when gypsum or molasses was added alone or in combination. The lowest spontaneously dispersed clay was found in the molasses+gypsum treatment, although this was not significantly different from the gypsum-only and molasses-only treatments, and the highest one was found in the control. When the soil was mechanically dispersed, gypsum, alone or combined with molasses, dramatically reduced clay dispersion, whereas the effect of molasses alone was much less pronounced. The highest mechanically dispersed clay was found in the control and the lowest one was found in the gypsum-only treatment.

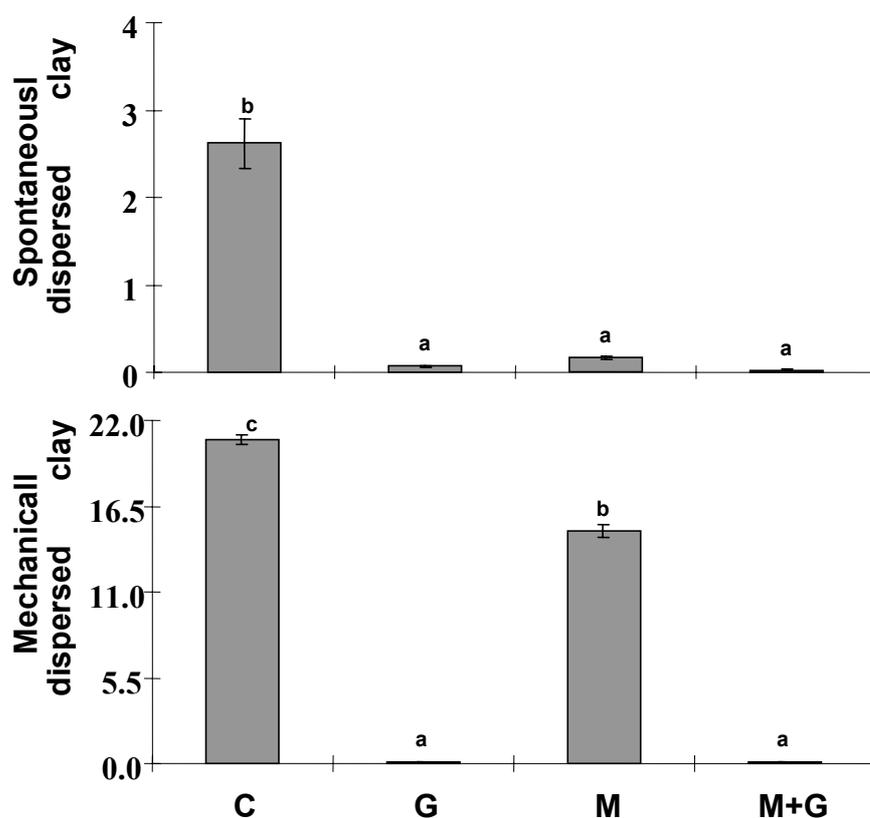


Figure 18 Spontaneous dispersion and mechanical dispersion of the Burdekin soil after gypsum and molasses treatments. C, M, G, M+G refer to control, molasses, gypsum and molasses+gypsum (each 10 t/ha), respectively. Letters show which results are significantly different ($P = 0.001$)

In the coarser-textured Proserpine soil (Figure 19), the amount of spontaneously and mechanically dispersed clay was generally much less than in the Burdekin soil. Spontaneous dispersion was reduced by all treatments, with molasses+gypsum being the most effective. The lowest spontaneously dispersed clay was found in the molasses+gypsum treatment and the highest was found in the control. When soil was mechanically disturbed, gypsum treatments produced a dramatic reduction in dispersed clay compared with the molasses-only treatment. The lowest mechanically dispersed clay was found in the gypsum-only treatment and the highest one was found in the control.

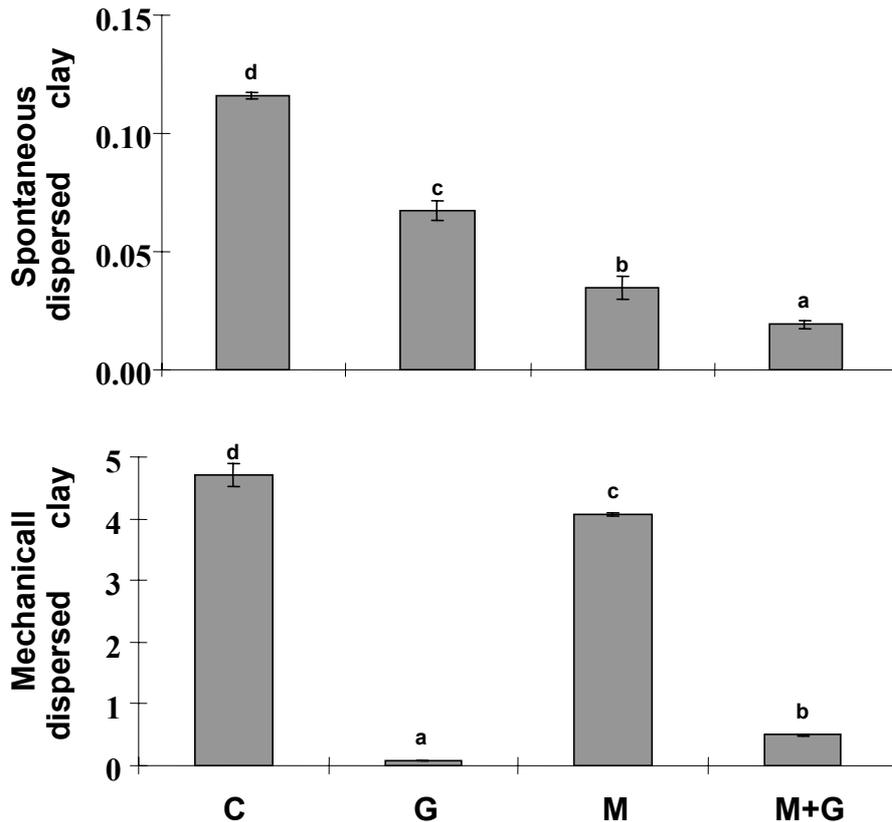


Figure 19 Spontaneous dispersion and mechanical dispersion of the Proserpine soil after gypsum and molasses treatments. C, M, G, M+G refer to control, molasses, gypsum and molasses+gypsum (each 10 t/ha), respectively. Letters show which results are significantly different ($P = 0.001$)

Wet aggregate stability

The effects of molasses and gypsum on wet aggregate stability of the soils are shown in Figure 20. In the Burdekin soil, molasses and gypsum treatments either alone or combined, provided significant increases in aggregate stability. Molasses and gypsum increased the proportion of macro-aggregates ($> 250 \mu\text{m}$) and decreased the proportion of aggregates $< 125 \mu\text{m}$, with the combined effects being greatest; aggregates in the range of $125\text{-}250 \mu\text{m}$ were relatively unaffected except by the combined treatment. In the Proserpine soil, gypsum had no significant effect on aggregate stability, either alone or

combined with molasses. Molasses significantly increased the proportion of macro-aggregates, while decreasing the proportion of both aggregate fractions $< 250 \mu\text{m}$.

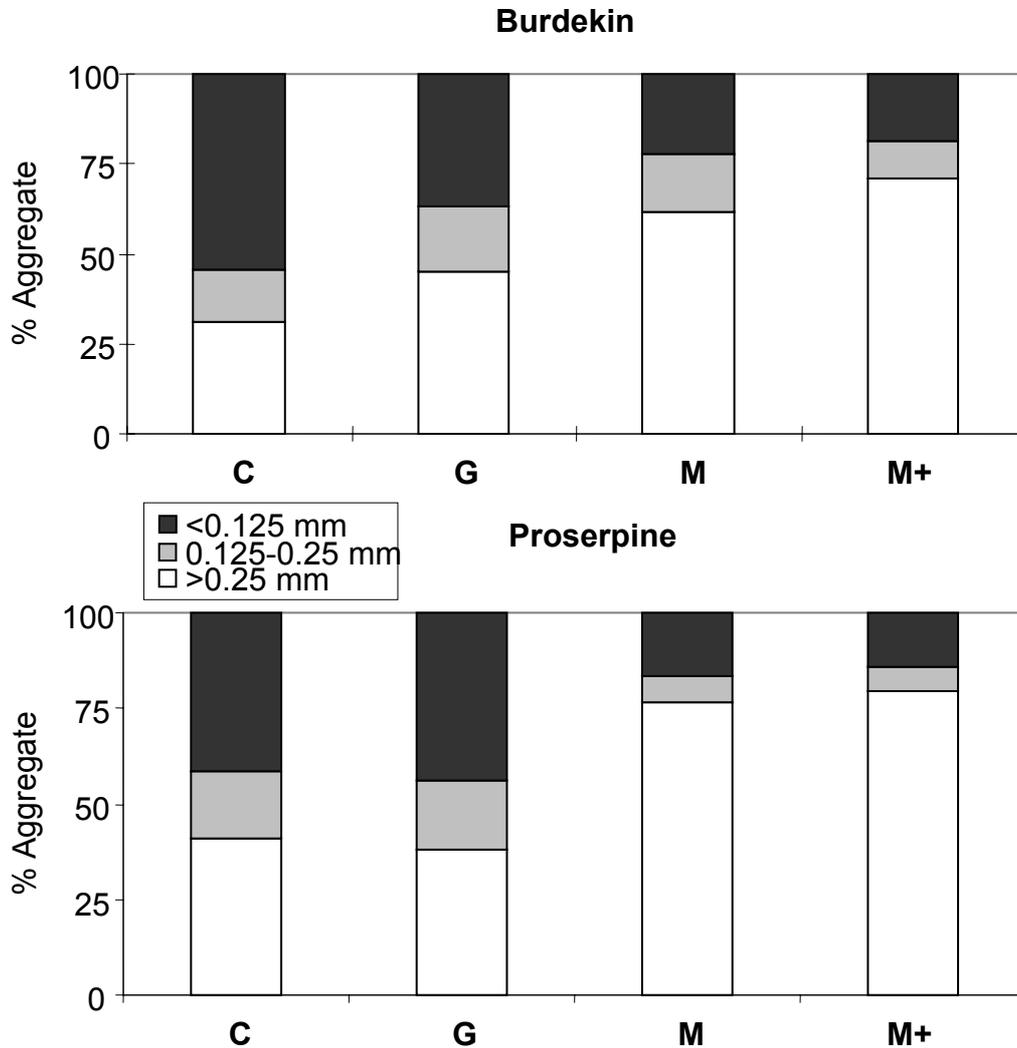


Figure 20 Wet aggregate stability of the Burdekin and Proserpine soils after molasses and gypsum treatments. C, M, G, M+G refer to control, molasses, gypsum and molasses+gypsum, respectively

Soil chemical properties

Table 26 shows the chemical properties of the two soils at the end of the incubation. Leaching alone decreased pH for the control treatment in the Burdekin soil, but the effect was reversed in the Proserpine soil. In the Burdekin soil, the decrease in pH was enhanced by the addition of both treatments with the gypsum-only treatment being greatest. However, molasses increased pH in the Proserpine soil, while gypsum did the opposite. The lowest pH was observed in the gypsum-only treatment for both soils and the highest pH was observed in the control and in the molasses-only treatment for the Burdekin and Proserpine soils, respectively.

The EC of both soils was significantly increased by gypsum but not by molasses (Table 26). In the Burdekin soil, the EC of the soil treated with molasses+gypsum was slightly greater than that treated with gypsum alone. The EC values of the gypsum-treated soils indicated that gypsum remained in the soil after five leaching events. The highest EC was observed in the molasses+gypsum treatment, and this was significantly different for all other treatments; the lowest EC was observed in the molasses-only treatment, although this was not significantly different from the control. In the Proserpine soil, the highest EC was observed in the gypsum-only treatment, although this was not significantly different from the molasses+gypsum treatment. The lowest EC was observed in the control, although this was not significantly from the molasses-only treatment.

Table 26 pH and electrical conductivity (EC) in 1:5 soil:water extracts, exchangeable sodium percentage (ESP), cation exchange capacity (CEC) and organic carbon content of the Burdekin and the Proserpine soils after molasses and gypsum treatments and five leaching events. C, G, M and M+G refer to control, gypsum (10 t/ha), molasses (10 t/ha) and molasses+gypsum, respectively

Treatment	pH	EC (dS/M)	ESP	CEC (cmol(+)/kg)	Org. C (%)
Burdekin soil					
Initial	8.0	0.2	7.9	10.9	0.8
C	7.7c	0.1a	4.1c	10.6a	0.8a
G	7.1a	1.6b	0.4a	10.4a	0.7a
M	7.6b	0.2a	2.4b	11.6b	1.0b
M+G	7.1a	1.9c	0.2a	11.2b	1.0b
Significance	*	*	**	**	**
Proserpine soil					
Initial	4.9	0.1	18.8	2.3	0.7
C	5.4b	0.1a	3.4c	2.1a	0.6a
G	4.6a	2.2b	0.3a	2.2a	0.7a
M	6.4d	0.1a	2.1b	2.5b	0.8b
M+G	6.1c	2.1b	0.2a	2.5b	0.9b
Significance	**	**	**	**	**

Letters (a, b, c, d) show which results are significantly different at P = 0.05 (*) and P = 0.01 (**)

Exchangeable sodium percentage (ESP) was determined to investigate the changes in sodicity after addition of molasses and gypsum; both treatments reduced the ESP substantially. Leaching alone decreased the ESP for the control in both soil and its effect was more pronounced in the Proserpine soil. In both soils, gypsum reduced ESP more than molasses because of the high concentration of calcium in solution. The lowest ESP was observed in the molasses+gypsum treatment although this not significantly different from those of gypsum-only. Cation exchange capacity (CEC) and organic carbon were significantly increased by molasses treatment in both soils, whereas the gypsum had no effect.

The concentrations of organic carbon in leachate and in soil were used to calculate the percentage of carbon lost in respiration, presumably released as CO₂ or leached as DOC during experiments (Table 27). In general, a substantial proportion of the molasses added

was lost in respiration (CO₂) but a small proportion was leached as DOC. In the molasses-only and molasses+gypsum treatments of both soils, the percentages of carbon released as CO₂ and leached as DOC were similar at 31-36% and 1-2%, respectively.

Table 27 Percentage of carbon lost in respiration (CO₂) and dissolved organic carbon (DOC) leached in the control (C), gypsum (G), molasses (M) and molasses+gypsum (M+G) treatments in the Burdekin and Proserpine soils

Treatment	Organic carbon (mg/kg)				% released as CO ₂	% leached as DOC
	Initial	Leachate (DOC)	Soil (final)	Leachate + soil		
Burdekin soil						
C	8000	73	8050	8123	-1	0.9
G	8000	33	7225	7258	9	0.5
M	14222	205	9525	9730	32	2.1
M+G	14222	164	9700	9864	31	1.7
Proserpine soil						
C	7000	23	6200	6223	11	0.4
G	7000	19	6700	6719	4	0.3
M	13222	175	8300	8475	36	2.1
M+G	13222	123	8625	8748	34	1.4

The concentrations of cations in the leachate and on the exchange sites of soil were used to calculate the abundances of cations in the system at the beginning and at the end of experiments (Table 28). Molasses and gypsum affected the proportion of cations in the leachate and on the exchange sites of both soils. In the Burdekin soil, when gypsum was added, Ca in the leachate and in the soil increased as expected. The displacement of other exchangeable cations by Ca can be seen in Table 27. In general, the proportions of Na, Mg and K cations in the leachate were higher than in the soil when gypsum was added.

Molasses also facilitated removal of Na, but not to the extent of the gypsum treatment. The proportions of Ca, Mg and K were generally higher in soil than in leachate for the molasses-only treatment in both soils; molasses contains considerable amounts of these cations. In the Burdekin soil, when molasses+gypsum was added, 95% of Na was removed in the leachate; in the Proserpine soil, a lower proportion of Na was found in the leachate. Adding molasses+gypsum partially compensated for the loss of K through leaching; this loss of plant available-K might be a negative effect of adding gypsum alone.

Table 28 Cation contents in soil (cmol(+)/kg) in control (C), gypsum (G), molasses (M) and molasses+gypsum (M+G) treatments

Treatment	Initial				Final				
	Soil	Addition	Total		Leachate	Soil	Total	Leachate (%)	Soil (%)
Ca, Burdekin									
C	6.5	0.0	6.5		0.1	7.1	7.2	1	99
G		14.4	20.9		1.9	17.9	19.8	9	91
M		0.04	6.6		0.2	7.2	7.4	3	97
M+G		14.4	20.9		1.9	19.9	21.7	9	91
Ca, Proserpine									
C	0.9	0.0	0.9		0.1	0.8	0.8	10	90
G		14.4	15.3		1.8	13.7	15.5	11	89
M		0.0	0.9		0.2	1.4	1.6	11	89
M+G		14.4	15.3		2.0	17.3	19.4	10	90
Mg, Burdekin									
C	3.9	0.0	3.9		0.1	4.8	4.9	3	97
G		0.0	3.9		3.0	1.3	4.3	69	31
M		0.1	4.0		0.3	4.7	5.0	6	94
M+G		0.1	4.0		3.4	1.6	4.9	68	32
Mg, Proserpine									
C	0.4	0.0	0.4		0.1	0.3	0.4	19	81
G		0.0	0.4		0.2	0.2	0.4	52	48
M		0.1	0.5		0.1	0.4	0.5	24	76
M+G		0.1	0.5		0.3	0.2	0.5	64	36
Na, Burdekin									
C	0.9	0.0	0.9		0.51	0.53	1.0	49	51
G		0.0	0.9		1.01	0.09	1.1	92	8
M		0.01	1.0		0.66	0.33	1.0	67	33
M+G		0.01	1.0		0.99	0.05	1.0	95	5
Na, Proserpine									
C	0.4	0.0	0.4		0.26	0.16	0.4	62	38
G		0.0	0.4		0.35	0.05	0.4	87	13
M		0.01	0.4		0.31	0.14	0.5	69	31
M+G		0.01	0.4		0.33	0.10	0.4	76	24
K, Burdekin									
C	0.5	0.0	0.5		0.02	0.5	0.5	5	95
G		0.0	0.5		0.3	0.01	0.3	96	4
M		0.2	0.7		0.1	0.8	0.9	7	93
M+G		0.2	0.7		0.7	0.1	0.8	89	11
K, Proserpine									
C	0.4	0.0	0.4		0.03	0.4	0.4	7	93
G		0.0	0.4		0.4	0.0	0.4	93	7
M		0.2	0.7		0.3	0.4	0.7	46	54
M+G		0.2	0.7		0.4	0.4	0.8	54	46

6.4.3 Discussion

The questions addressed in this study were whether molasses alone or combined with gypsum could improve the structural stability of sodic soils. The results of leachate analysis show that additions of molasses and gypsum significantly changed the properties of sodic topsoils. For the soil leachates, gypsum increased EC and decreased pH immediately. The effect of the gypsum on pH was due to the displacement of adsorbed protons by calcium. Similar results were found by Sekhon and Bajwa (1993) and by Chorom and Rengasamy (1997). However, this effect may be temporary and progressively lost as the gypsum is leached.

In the Burdekin soil, the EC of leachate in the molasses+gypsum treatment was generally lower than of the treatment with gypsum alone (Figure 16). This may indicate that some soluble calcium is complexed by organic matter and immobilized in the soil. Conversely, organic matter may have been protected from leaching by the presence of soluble Ca in the soil. This can be seen in Figure 17 and Table 27, where DOC was generally lower with molasses+gypsum than with molasses alone. Muneer and Oades (1989) also found that DOC decreased when gypsum and glucose were added to soil. When gypsum was applied, the concentration of DOC decreased; soluble calcium from gypsum inhibited decomposition of organic matter by microorganisms and reduced the release of the DOC by leaching. Calcium tends to block and protect functional groups in organic matter that represent sites of initial decomposition and also cross-links flexible polymers to create more dense, rigid molecules that are more stable to both chemical and biological degradation (Oades 1989). However, in the Proserpine soil the EC after molasses+gypsum treatment was higher than after gypsum alone (Figure 17). This may be because the clay content of the soil was too low to protect cations in the soil solution from leaching so that the effects of molasses and gypsum on EC were simply additive in this case.

The small decrease in pH of the Burdekin soil brought about by addition of molasses is most likely due to the pH of the molasses (5.5), and to the CO₂ and organic acids produced during incubation. However, in the less buffered, more acidic Proserpine soil, addition of molasses increased soil pH to a higher value than that of either the soil or the molasses. This may have been due to reactions such as the mineralisation of cations or nitrogen in the molasses, or to the uptake of nitrate by microorganisms.

Cation exchange capacity (CEC) in both soils increased significantly with molasses but not with gypsum treatment. This was presumably due to the decomposition products of the carbohydrates in molasses having high CEC. The data in Table 26 suggest that this additional organic matter in the Burdekin soil had a CEC of approximately 300 cmol(+)/kg C, whereas that in the Proserpine soil had a CEC of about 160 cmol(+)/kg C; these values were calculated from the increases in soil CEC in relation to increases in organic carbon (Table 26). Organic exchange sites may have a higher selectivity for Ca than mineral exchange sites (Nelson and Oades 1998).

The water stability of aggregates and the clay dispersibility of the soils were significantly affected by molasses and gypsum. In the Burdekin soil, both molasses and gypsum treatments increased macro-aggregation at the expense of micro-aggregates. Organic matter from molasses, probably mostly carbohydrate, was able to stabilise micro-

aggregates into macro-aggregates, while the gypsum effect was mainly due to the flocculation of clay resulting in the aggregation of clay particles into domains and reduced dispersion (Chorom and Rengasamy 1997; Nelson and Oades 1998). In the Proserpine soil, molasses increased wet aggregate stability to an even greater degree than in the Burdekin soil, but, as with the Burdekin soil, had a far less dramatic effect on mechanical dispersion, indicating that molasses was unable to stabilise aggregates under these more destructive conditions. The resistance to slaking afforded by increased organic matter is probably insufficient to withstand the rigours of mechanical dispersion in which the preliminary destruction of macro-aggregates provides opportunities for extensive dispersion. On the other hand, gypsum was able to prevent clay from dispersing but unable to maintain macro-aggregates. The extents of both spontaneous and mechanical dispersion in both soils generally reflect the ESP and EC data in Table 26 with higher ESP and lower EC promoting dispersion. It is significant that treatment of both soils with molasses alone resulted in substantial reductions in ESP and spontaneous dispersion.

6.5 Structural stability of sodic soils as influenced by molasses and lime

The addition of gypsum to soil can increase the stability of aggregates by lowering the exchangeable sodium percentage (ESP), increasing the electrolyte concentration and, thereby, reducing dispersion. However, gypsum additions are lost relatively quickly when water infiltrates through soil because gypsum is quite soluble. Lime is a less soluble source of Ca, and also tends to be cheaper than gypsum in the Queensland sugarcane-growing districts. Lime may in fact be too insoluble to be of benefit, particularly in soils with high pH. However, when considering the joint use of lime and molasses, there are several mechanisms whereby the addition of molasses may help to dissolve the lime. Organic acids and CO₂ are produced by microbial decomposition of organic matter, and under certain circumstances may reduce soil pH (Baldock *et al.* 1994). Therefore, in soils of high pH, the addition of large quantities of organic matter in the form of molasses might be expected to lower pH out of the alkaline range. The increased partial pressure of CO₂ and any organic acid might then aid the dissolution of the lime, freeing Ca²⁺ ions (Robbins 1986) and lowering the soil pH. The Ca²⁺ ions may then exchange with Na⁺ ions adsorbed to clay particles, allowing Na⁺ ions to be leached, thus eventually decreasing the ESP (Lehrsch *et al.* 1993).

In this trial, we assess the influence of molasses and lime on the structural stability of soils.

6.5.1 Materials and methods

The materials and methods for this experiment are virtually identical to those described in Section 6.4.1, except that the treatments in this experiment were molasses and lime rather than molasses and gypsum. The amount of calcium added to the soils as lime was similar to that added as gypsum described in the previous experiment (Section 6.4.1), and the application rate of molasses was also the same. Table 1 describes some of the properties of the soils we used.

6.5.2 Results

Leachate properties of treated soils

Figures 21 and 22 show the effects of molasses and lime on the properties of five successive leachates from the Burdekin and Proserpine soils. The first leaching was conducted on soil that had been slowly wetted to 85% of field capacity and kept for 7 d.; subsequent leaching events were conducted after soil had been air-dried from the previous leaching event.

There were significant changes in leachate properties related to molasses and lime treatments. In general, pH, electrical conductivity (EC), sodium adsorption ratio (SAR) and dissolved organic carbon (DOC) decreased as leaching proceeded, although the magnitude of reductions fluctuated. In the Proserpine soil, the pH of the first leachate was lower than that of the second leachate in all treatments, even though these were not significantly different for the lime-only and the molasses+lime treatments. In the Burdekin soil, by contrast, the pH was lower in the first leachate of the control and the molasses-only treatments. This may simply be an experimental artefact caused by the first leaching being conducted on moist soil, while subsequent leachings were conducted on air-dried soil. Similar results were found by Nelson *et al.* (1998), where the pH of soil is lower under moist conditions than dry. The average pH of the Burdekin leachates ranged between 7.2 and 8.2 in all treatments, while for the Proserpine soil the range was 4.8 to 8.0. In the Burdekin soil, the highest and the lowest pH were observed in the first leachate of the lime-only and the control treatments respectively and in the Proserpine, the highest and the lowest pH were observed in the second leachate of the molasses+lime and the first leachate of the control, respectively.

Electrical conductivity (EC) of leachates decreased from the first to the fifth leachate in virtually all treatments for both soils. This indicates that cation dissolution was greater in the early stages and diminished as successive leaching proceeded. The EC of each leachate in the molasses+lime treatment was usually higher than for the leachates of all other treatments in the same leaching events.

Sodium adsorption ratio (SAR) of leachates decreased as leaching proceeded in all treatments for both soils. In the Burdekin soil, the treatments had little effect on leachate SAR, whereas in the Proserpine soil SAR values were highest in the molasses and lime+molasses treatments.

In all leaching events, dissolved organic carbon (DOC) concentration was highest in the molasses-only treatment and second highest in the lime+molasses treatment. It was highest in the first or second leaching event in all cases, and decreased thereafter. In the Burdekin soil, lime alone decreased DOC, whereas in the Proserpine soil lime alone had no effect.

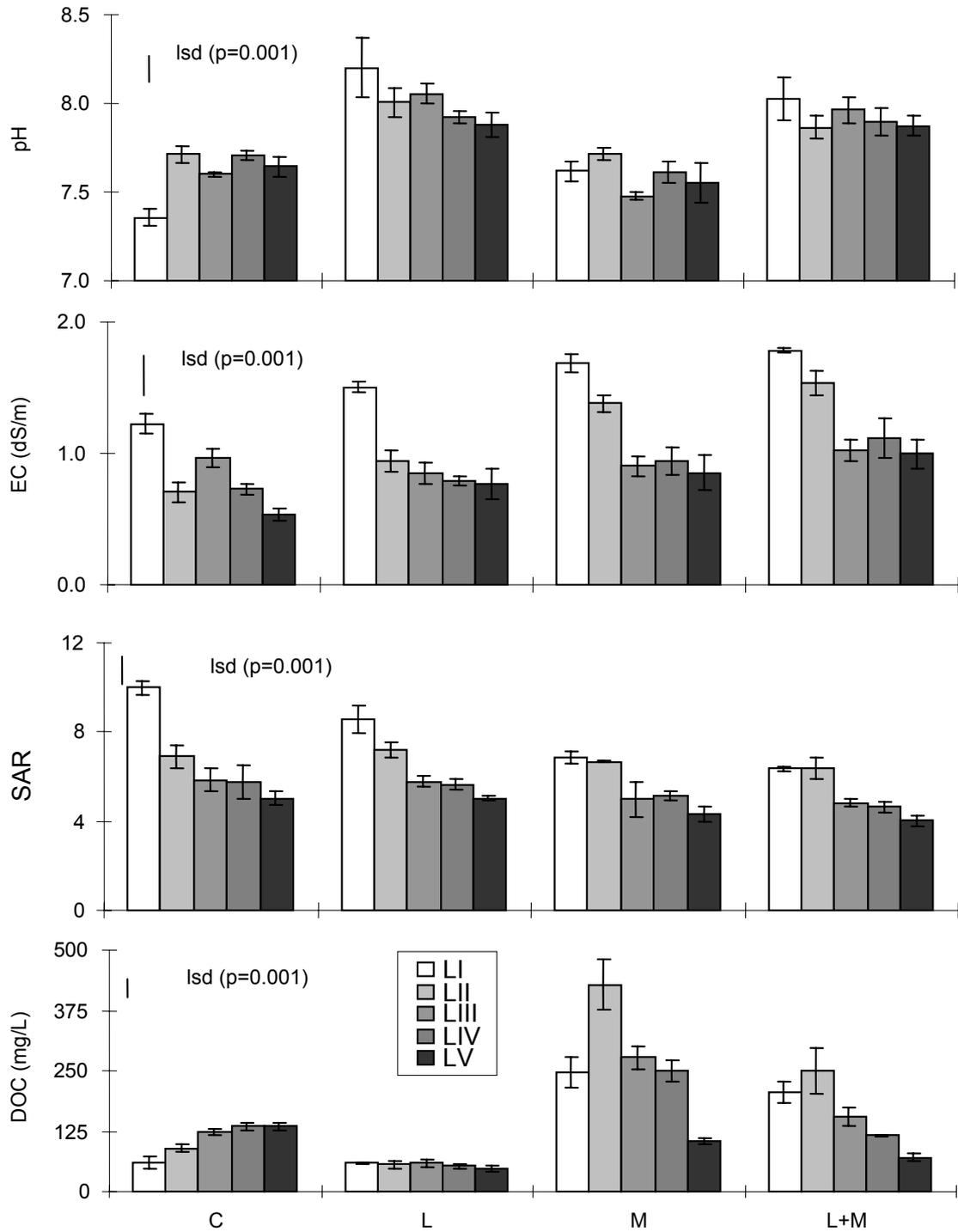


Figure 21 Burdekin soil: pH, electrical conductivity (EC), sodium adsorption ratio (SAR) and dissolved organic carbon (DOC) of five successive leaching events (LI- LV) after molasses and lime treatments. C, L, M and M+L refer to control, lime, molasses, molasses + lime. Error bars represent one standard deviation

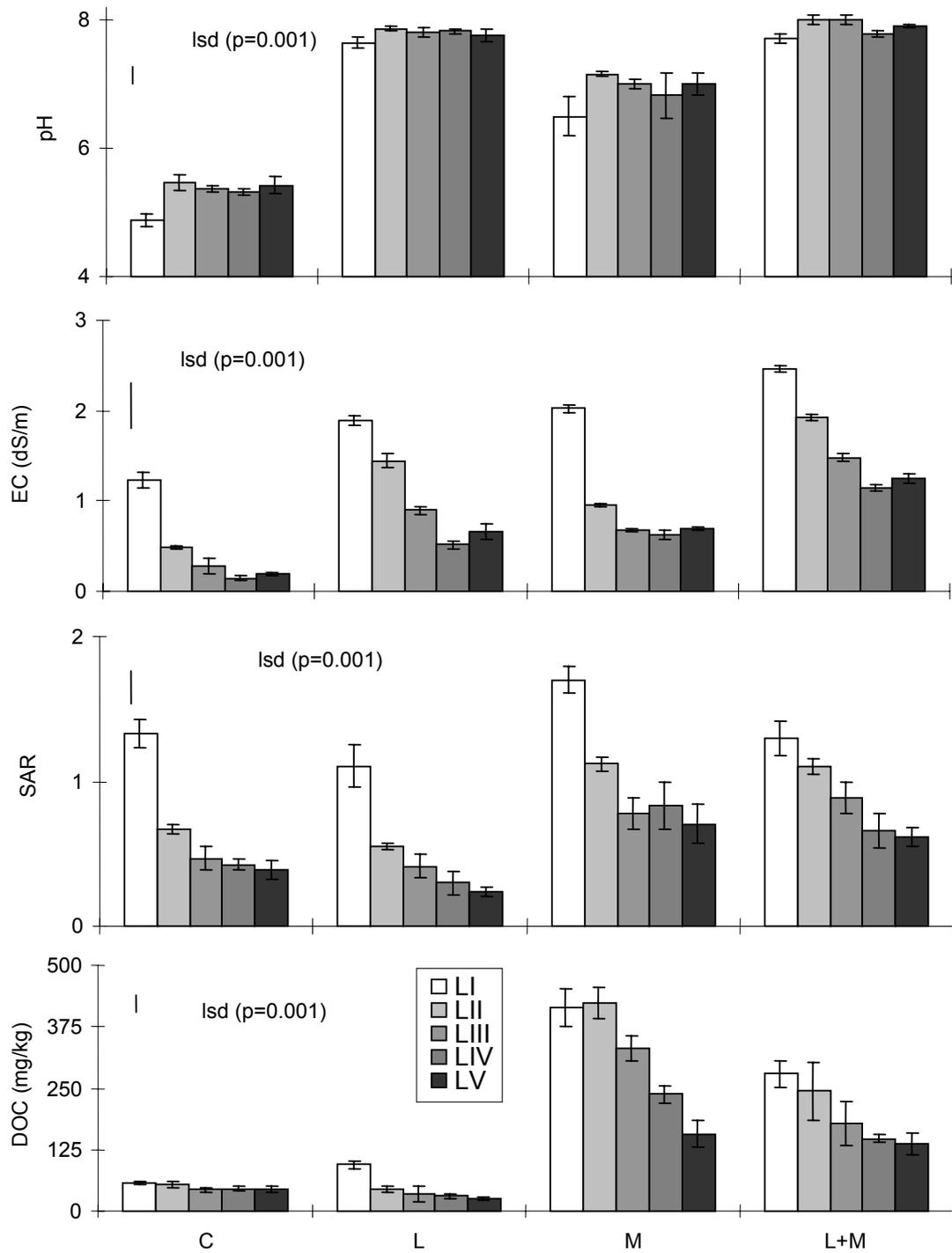


Figure 22 Proserpine soil: pH, electrical conductivity (EC), sodium adsorption ratio (SAR) and dissolved organic carbon (DOC) of five successive leaching events (LI- LV) after molasses and lime treatments. C, L, M and M+L refer to control, lime, molasses, molasses + lime. Error bars represent one standard deviation

Clay dispersibility

Figure 23 and 24 present the change in spontaneous and mechanical dispersion in the soils. In the Burdekin soil, molasses and lime, alone or together, decreased spontaneous dispersion dramatically and to an equal extent (Figure 23). The amount of dispersed clay for each treatment was markedly greater when the soil was mechanically dispersed. Lime, molasses and lime+molasses treatments significantly reduced ($P = 0.01$) the dispersed clay compared with the control.

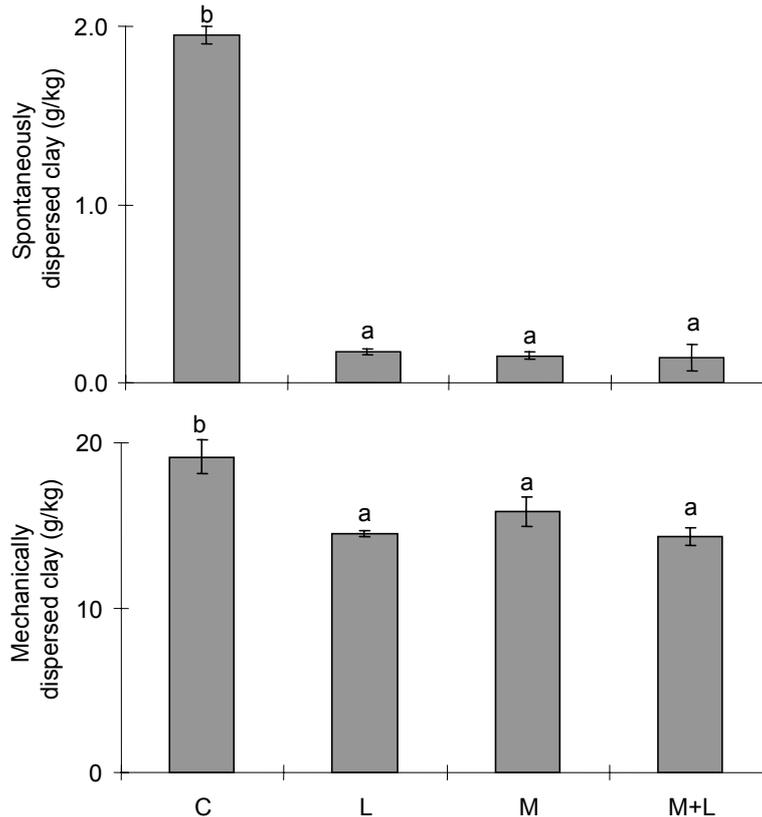


Figure 23 Spontaneous and mechanical dispersion of the Burdekin soil as influenced by molasses and lime. C, L, M, M+L represent control, lime-only, molasses-only and molasses+lime treatments, respectively. Letters show results that are significantly different ($P = 0.01$). Error bars represent one standard deviation

In the coarser-textured Proserpine soil, as was observed in the previous experiment (Section 6.4), the amount of dispersed clay in spontaneous and mechanical dispersion was much less than that in the Burdekin soil (Figure 24). The spontaneously dispersed clay was significantly higher in the control than in the other treatments. However, when the soil was mechanically dispersed, the amounts of the dispersed clay in the molasses-only and lime-only treatments were not significantly different from the control.

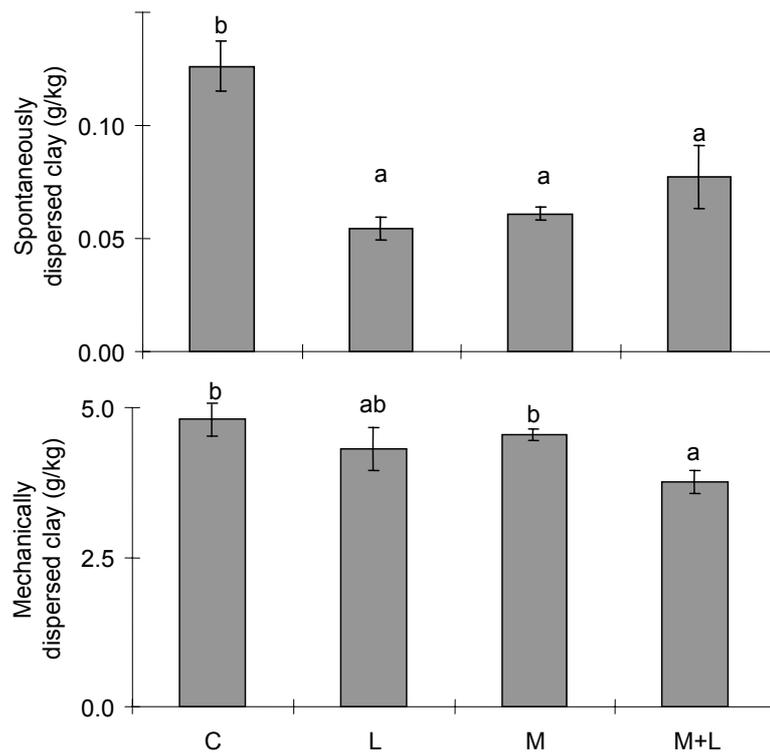


Figure 24 Spontaneous and mechanical dispersion of the Proserpine soil as influenced by molasses and lime. C, L, M, M+L represent control, lime-only, molasses-only and molasses+lime treatments, respectively. Letters show results that are significantly different ($P = 0.01$). Error bars represent one standard deviation

Wet aggregate stability

Figure 25 shows the influence of molasses and lime application on wet aggregate stability of soils. In the Burdekin soil, molasses-only, lime-only and molasses+lime treatments all increased the proportion of macroaggregates (> 0.250 mm) largely at the expense of aggregates fraction < 0.125 mm. This indicates an increase in water stability of aggregates. The greatest proportion of macroaggregates (or the smallest proportion of aggregates < 0.125 mm) occurred in the molasses+lime treatment, although this was not significantly different ($P = 0.01$) from the molasses-only treatment, yet significantly different from the lime-only treatment. The proportion of 0.125 - 0.250 mm aggregates was relatively constant in all treatments. In the Proserpine soil, the proportion of macroaggregates increased at the expense of both smaller aggregate fractions in both molasses-only and molasses+lime treatments. In the lime-only treatment, the proportion of aggregates < 0.125 mm appears to have increased marginally with a complementary decrease in aggregates 0.125 - 0.250 mm, while the proportion of macroaggregates remained unchanged.

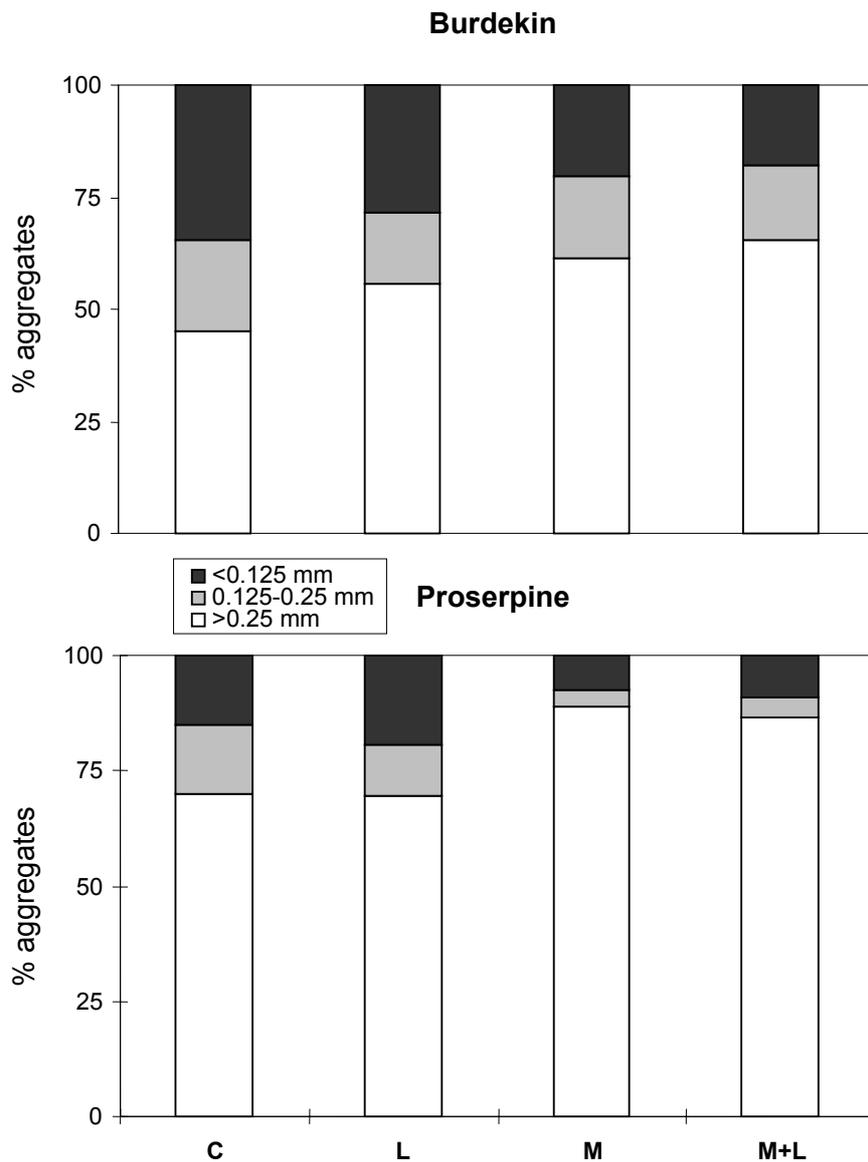


Figure 25 Wet aggregate stability of soils as influenced by molasses and lime treatments. C, L, M, M+L represent control, lime-only, molasses-only and molasses+lime treatments, respectively

Chemical properties of treated soils

Table 29 presents the change in chemical properties of the Burdekin and Proserpine soils after molasses and lime treatments. In the Burdekin soil, pH decreased in the control and with molasses alone, increased with lime alone, and stayed the same with lime+molasses. In Proserpine soil, pH increased in all treatments.

The EC of both soils was generally increased by molasses and lime applications, with their effects being additive. Leaching alone reduced the EC for both soils with the effect being more pronounced in the Proserpine soil. The EC for both soils increased in the order of treatments control <lime <molasses <molasses+lime.

Table 29 pH and electrical conductivity (EC) in 1:5 soil:water extracts, exchangeable sodium percentage (ESP), cation exchange capacity (CEC) and organic carbon content of the Burdekin and the Proserpine soils after molasses and gypsum treatments and five leaching events. C, L, M and M+L refer to control, lime (10 t/ha), molasses (10 t/ha) and molasses+lime (both 10 t/ha), respectively.

Treatment	pH	EC (dS/m)	ESP	CEC (cmol(+)/kg)	Org. C (%)
	(1:5 soil:water)				
Burdekin soil					
Initial	8.0	0.16	7.9	10.9	0.8
C	7.7 b	0.06 a	4.3 c	10.6 a	0.81 a
L	8.5 d	0.14 b	2.0 a	10.4 a	0.83 a
M	7.5 a	0.16 b	3.4 b	11.6 b	0.95 b
M+L	8.1 c	0.27 c	1.5 a	11.2 b	0.91 b
Significance	**	**	**	**	**
Proserpine soil					
Initial	4.9	0.12	18.0	2.3	0.7
C	5.4 a	0.06 a	7.3 c	2.1 a	0.62 a
L	8.1 d	0.10 b	0.9 a	2.2 a	0.71 a
M	6.0 b	0.13 c	4.6 b	2.5 b	0.83 b
M+L	7.5 c	0.21 d	1.0 a	2.5 b	0.86 b
Significance	**	**	**	**	**

Letters show results significantly different at P = 0.01 (**)

The exchangeable sodium percentage (ESP) was reduced by molasses and lime treatments in both soils. Leaching alone (with distilled water) nearly halved the ESP in the Burdekin soil and more than halved it in the Proserpine soil. This result was consistent with the previous experiment (Section 6.4). The effect of lime-only treatment in reducing the ESP appears to be more pronounced in the Proserpine soil than that in the Burdekin soil. In both soils, final ESP decreased in the order control >molasses >lime =molasses+lime.

The cation exchange capacity (CEC) and organic carbon content were significantly increased for both soils after addition of molasses but not lime.

The abundance of cations in the system at the beginning and at the end of the experiment is shown in Table 30. In general, molasses and lime increased the proportion of cations in the leachate and on exchange sites of both soils. The cations in the leachates for each treatment were higher in the Proserpine soil than that in the Burdekin soil except for sodium. This reflects the coarser texture of Proserpine soil relative to that of the Burdekin soil. It should be noted, however, that the volume of water leached during each event was greater in the Burdekin soil than in the Proserpine soil (88 mL versus 72 mL). The proportion of cations in leachate was small compared with those on the exchange phases of soil in all treatments except sodium in the Burdekin soil. In the Burdekin soil, it appeared that Ca from lime displaces exchangeable Na but not exchangeable Mg and K, which is fortunate for plant nutrition.

Table 30 Cation abundance (cmol(+)/kg) in control (C), lime (L), molasses (M) and molasses+lime (M+L) treatments

Treatment	Initial			Final				
	Soil	Addition	Total	Leachate	Soil	Total	Leachate (%)	Soil (%)
Ca, Burdekin								
C	7.9	0.0	7.9	0.1	7.4	7.5	1.2	98.8
L		17.9	25.8	0.1	23.3	23.4	0.6	99.4
M		0.04	7.9	0.2	7.5	7.7	2.4	97.6
L+M		17.9	25.8	0.2	14.5	14.7	1.5	98.5
Ca, Proserpine								
C	0.9	0.0	0.9	0.1	0.8	0.9	9.1	90.9
L		17.9	18.7	0.5	15.8	16.3	3.3	96.7
M		0.04	0.9	0.1	0.9	1.0	11.2	88.8
L+M		17.9	18.8	0.5	15.0	15.5	3.2	96.8
Mg, Burdekin								
C	4.7	0.0	4.7	0.1	4.8	4.9	1.9	98.1
L	4.7	0.0	4.7	0.1	4.3	4.4	2.4	97.6
M		0.1	4.8	0.2	4.7	4.9	4.3	95.7
L+M		0.1	4.8	0.2	4.2	4.4	5.0	95.0
Mg, Proserpine								
C	0.4	0.0	0.4	0.1	0.3	0.4	28.3	71.7
L	0.4	0.0	0.4	0.1	0.3	0.4	26.0	74.0
M		0.1	0.4	0.1	0.6	0.7	18.1	81.9
L+M		0.1	0.4	0.2	0.6	0.8	25.6	74.4
Na, Burdekin								
C	1.4	0.0	1.4	0.5	1.2	1.7	30.1	69.9
L	1.4	0.0	1.4	0.6	0.6	1.2	49.6	50.4
M		0.01	1.4	0.6	0.5	1.2	54.4	45.6
L+M		0.01	1.4	0.6	0.4	1.0	61.2	38.8
Na, Proserpine								
C	0.7	0.0	0.7	0.1	0.7	0.7	8.0	92.0
L	0.7	0.0	0.7	0.1	0.7	0.8	11.4	88.6
M		0.01	0.7	0.1	0.7	0.8	11.6	88.4
L+M		0.01	0.7	0.1	0.7	0.8	16.6	83.4
K, Burdekin								
C	0.5	0.0	0.5	0.01	0.5	0.5	1.3	98.7
L	0.5	0.0	0.5	0.01	0.5	0.5	1.1	98.9
M		0.2	0.8	0.05	0.9	0.9	5.1	94.9
L+M		0.2	0.8	0.05	0.8	0.9	5.3	94.7
K, Proserpine								
C	0.4	0.0	0.4	0.03	0.4	0.4	6.7	93.3
L	0.4	0.0	0.4	0.04	0.4	0.4	9.3	90.7
M		0.2	0.7	0.3	0.6	0.9	31.8	68.2
L+M		0.2	0.7	0.4	0.5	0.9	42.2	57.8

The proportion of carbon lost in respiration, presumably released as CO₂, and leached as DOC was calculated to show the balance of organic carbon in the system during the period of experiment (Table 31). The different amounts of carbon in the soil (before and after incubation) and in the leachate were assumed to be due to carbon lost in respiration. In general, a substantial proportion of molasses added was lost in respiration (CO₂) and only a small proportion was leached as DOC. In the molasses-only treatment, the proportion of CO₂ was higher in the Proserpine soil than that in the Burdekin soil. In both soils, the proportions of carbon lost in respiration and leached as DOC in the molasses-only and molasses+lime treatments were in range of 30-36% and 1-2%, respectively. DOC was higher in the molasses-only than in the molasses+lime treatments, which is consistent with what was observed in the previous pot trial (Section 6.4). In the lime-only and control treatments, the proportion of carbon lost in respiration was very small compared with molasses-only and molasses+lime treatments.

Table 31 Percentage of carbon lost in respiration (CO₂) and dissolved organic carbon (DOC) leached in the control (C), lime (L), molasses (M) and molasses+lime (M+L) treatments in the Burdekin and Proserpine soils

Treatment	Organic carbon (mg/kg soil)				% released as CO ₂	% leached as DOC
	Initial	Leachate (DOC)	Soil (final)	Leachate + soil		
Burdekin soil						
C	7800	96	8050	8146	-4	1.2
L	7800	48	8303	8351	-7	0.6
M	14022	230	9525	9755	30	1.6
M+L	14022	141	9146	9287	34	1.0
Proserpine soil						
C	7000	35	6200	6235	11	0.5
L	7000	33	7136	7169	-2	0.5
M	13222	227	8300	8527	36	1.7
M+L	13222	140	8621	8761	34	1.1

6.5.3 Discussion

Lime and molasses both had significant effects on soil properties. The addition of molasses did not increase the solubility of lime as was expected, despite a significant amount of CO₂ being produced. Good aeration may have prevented a build-up of CO₂ concentration in the soil.

DOC of the leachates increased markedly in the molasses treatments. However, when lime was also added, the DOC was lower than that of the molasses-only treatment. This may indicate that some dissolved organic matter was protected from leaching by the presence of calcium in the soil, as was found and discussed in the previous pot trial (Section 6.4).

Further experiments are needed to determine the best combination rates of molasses and lime to overcome problems associated with sodicity. In the Proserpine soil, the pH was increased when molasses was added as was observed and discussed in pot trial 2.

Wet aggregate stability and clay dispersibility of soils were influenced by molasses and lime. An increase in macroaggregation occurred at the expense of microaggregation, which may have been due to carbohydrate production after molasses addition plus fungal hyphae produced during incubation. Observations made during the experiments indicate that fungal hyphae were abundant in the soils to which molasses was added. However, macroaggregates are easily disrupted and broken into microaggregates when increasing dispersive energy was applied to soil. In the Burdekin soil, which had a clay content 4 times higher than the Proserpine soil, the amount of spontaneously and mechanically dispersed clay brought about by the molasses and lime treatments were significantly lower than that of the control. This probably results from strong binding of mucilages and colloidal organo-mineral complexes, which may have induced less clay to disperse even though the higher dispersive energy was applied to soil (Tisdall and Oades 1982).

6.6 Effect of trash management on susceptibility of soils to becoming or remaining sodic

Depending on management, soils can become more or less sodic. For example, irrigation with saline water increases soils sodicity, as does rising groundwater. On the other hand, application of gypsum or lime, together with irrigation with good quality water and adequate drainage, can decrease sodicity. For any given management regime, the extent to which soil becomes more or less sodic depends on the 'cation exchange selectivity' of the soil.

Cation exchange selectivity refers to the fact that some cations are held more strongly than others. For example, the cations Ca^{2+} and Mg^{2+} , which have two charges (divalent), are held more strongly than Na^+ and K^+ , which have one charge (monovalent). However, the difference between binding strength of divalent and monovalent cations depends on the nature of the cation exchange sites. Exchange sites on organic matter generally have higher relative selectivity for divalent cations than exchange sites on clays. Therefore, soils with high organic matter content are generally less susceptible to becoming sodic than soils with low organic matter content (Nelson and Oades 1998).

The hypothesis tested in this experiment was that sugarcane trash management, through its effect on soil organic matter, alters the cation exchange selectivity of soil. Therefore, GCTB, as opposed to burning, may make soils less susceptible to becoming sodic, and easier to ameliorate if already sodic.

6.6.1 Materials and method

Soils were sampled from the long-term GCTB/burnt trials at Ayr, Mackay and Tully (0-5 cm depth) and from a long-term pasture/wheat-fallow trial in Adelaide (0-10 cm depth). Soil properties are shown in Table 32.

Table 32 Selected properties of soil used in this experiment

Soil sites and management	OC (g/kg)	CEC (cmol(+)/kg)	pH	Particle size (%)			Mineralogy
				Clay	Silt	Sand	
Ayr (B)	14.4	8.0	7.04	10	11	79	I#, K#, Q*, F*
Ayr (G)	17.5	9.5	6.91				
Tully (B)	12.3	7.3	5.53	34	35	31	K#, S#, I*, Q*, F*
Tully (G)	18.1	11.0	5.23				
Mackay (B)	13.1	7.7	5.49	18	25	57	K#, F#, Q*, I*
Mackay (G)	14.3	7.7	5.35				
Urrbrae (WF)	11.4	9.9	5.85	11	27	62	RIM#, I#, Q*, K*
Urrbrae (PP)	31.1	10.3	5.65				

OC = organic C content, CEC = cation exchange capacity, pH is in 1:5 soil:water extract, K = kaolinite, RIM = randomly interstratified minerals, I = illite, Q = quartz, S = smectite, F = feldspar, B = burnt, G = green, WF = wheat/fallow and PP = permanent pasture. * and # = minor and major clay mineral

Cation exchange selectivity experiments are usually conducted by mixing the soils with solutions containing various salt solutions, shaking them, and then removing the solution by centrifugation. That technique, while rapid and simple, destroys soil structure and does not mimic the percolation of water that occurs in the field. In this experiment, soil aggregates were placed on sintered glass funnels, and solutions were introduced and removed by slowly raising or lowering the pressure head. The technique, while slow and laborious, is a much better representation of the conditions under which cation exchange occurs in the field. Mercury (1 mg/L) was added to all salt solutions to prevent microbial activity during the experiment.

The soils (three replicates of each) were initially saturated with CaCl₂, and excess salt was removed with water washes. They were then equilibrated, by repeated saturation and drainage, with solutions containing different amounts and proportions of NaCl and CaCl₂. In the solutions, total amount of salt is expressed as total cation concentration (TCC), and the amount of sodium as a proportion of the total salt is expressed as sodium adsorption ratio (SAR). Two levels of TCC (2 and 10mmol(+)L⁻¹) and two levels of SAR (3 and 10 mmol(+)L⁻¹) were used. Higher values of TCC and SAR could not be used because they led to aggregate collapse and clogging of the sintered funnel with dispersed clay.

After equilibration, solutions were analysed for Na and Ca using flame photometry and atomic absorption spectrophotometry. Cation exchange selectivity, expressed as the Gapon coefficient (k_G , see equation below), was calculated using mass balances. This method was validated by sampling the soils and measuring exchangeable cation contents. The higher the Gapon coefficient, the higher the selectivity for Na, and the more likely the soil is to become or remain sodic.

$$\text{Gapon coefficient } (k_G) = \text{ESR} / \text{SAR}$$

Where ESR or exchangeable sodium ratio is a measure of the sodicity of the soil (exchangeable Na divided by exchangeable Ca), similar to ESP, and where SAR or sodium adsorption ratio is the sodicity of the water passed through the soil.

6.6.2 Results and discussion

Figure 26 shows that the method developed for measuring k_G , using solution properties and a mass balance approach, gave virtually identical results to the conventional method, using soil and solution properties.

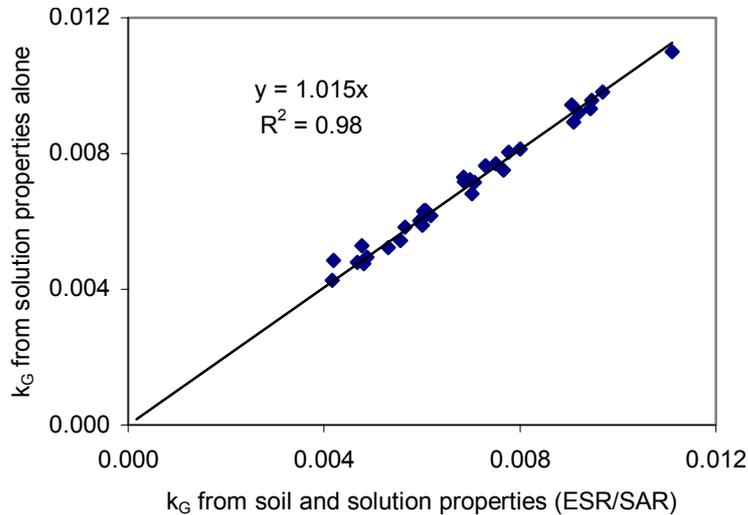


Figure 26 Relationship between k_G from soil and solution properties (ESR/SAR) and from solution properties alone

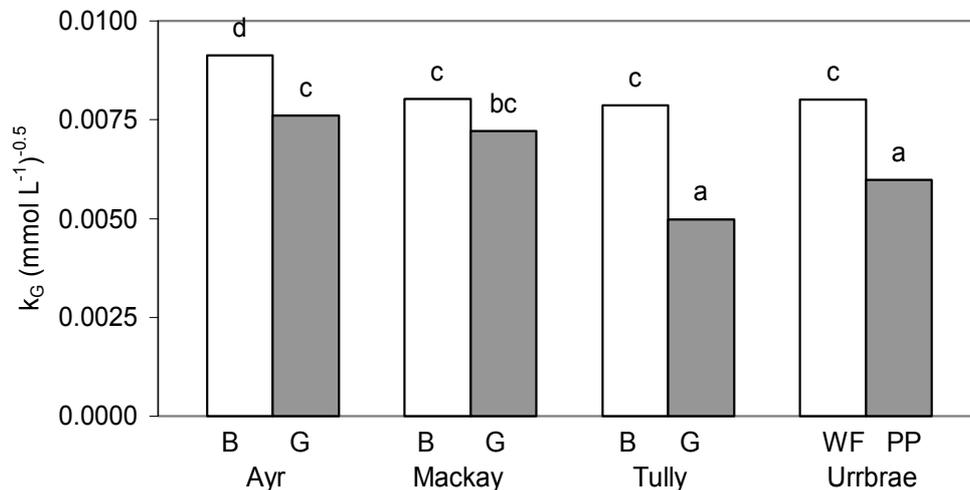


Figure 27 Effects of soil type and trash management practices (B burnt, G GCT, WF wheat/fallow, PP permanent pasture) on the average value of k_G (irrespective of TCC and SAR treatments)

Figure 27 shows the effect of soil type and sugarcane trash management practices (organic matter content) on the average value of k_G (mean for all levels of TCC and SAR). Analysis of variance showed that there was a highly significant effect ($P < 0.001$) of soil

type and trash management practices on values of k_G (Table 33). There was also a significant interaction effect ($P = 0.05$) between soil type and trash management on the value of k_G .

Table 33 Significance of the treatments as indicated by analysis of variance

Effect	k_G
Trash management (C)	**
SAR (N)	*
TCC (T)	ns
Soil (S)	**
C x N	ns
C x T	ns
N x T	**
C x S	*
N x S	*
T x S	ns
C x N x T	*
C x N x S	ns
C x T x S	ns
N x T x S	ns
C x N x T x S	*

* and ** indicate significant at $P = 0.05$ and $P = 0.001$, respectively
ns = non significant

Soil type significantly ($P = 0.001$) influenced the value of k_G , probably due to clay content and type. The average values of k_G decreased in the order Ayr (0.0084), Mackay (0.0076), Urrbrae (0.0070) and Tully (0.0064 (mmol L^{-1})^{-0.5}). Trash management practices significantly ($P = 0.001$) influenced the value of k_G . Figure 27 shows that the green treatment generally resulted in lower k_G values than the burnt treatment, even though the organic matter contents resulting from these management practices (Table 8) were not significantly different in Mackay. SAR significantly ($P = 0.05$) affected the value of k_G while TCC did not (Table 33). However, the interaction between SAR and TCC significantly influenced ($P = 0.001$) the value of k_G (Figure 28). In general, the value of k_G decreased as SAR increased. These trends were generally the same for all soils within each trash management practice.

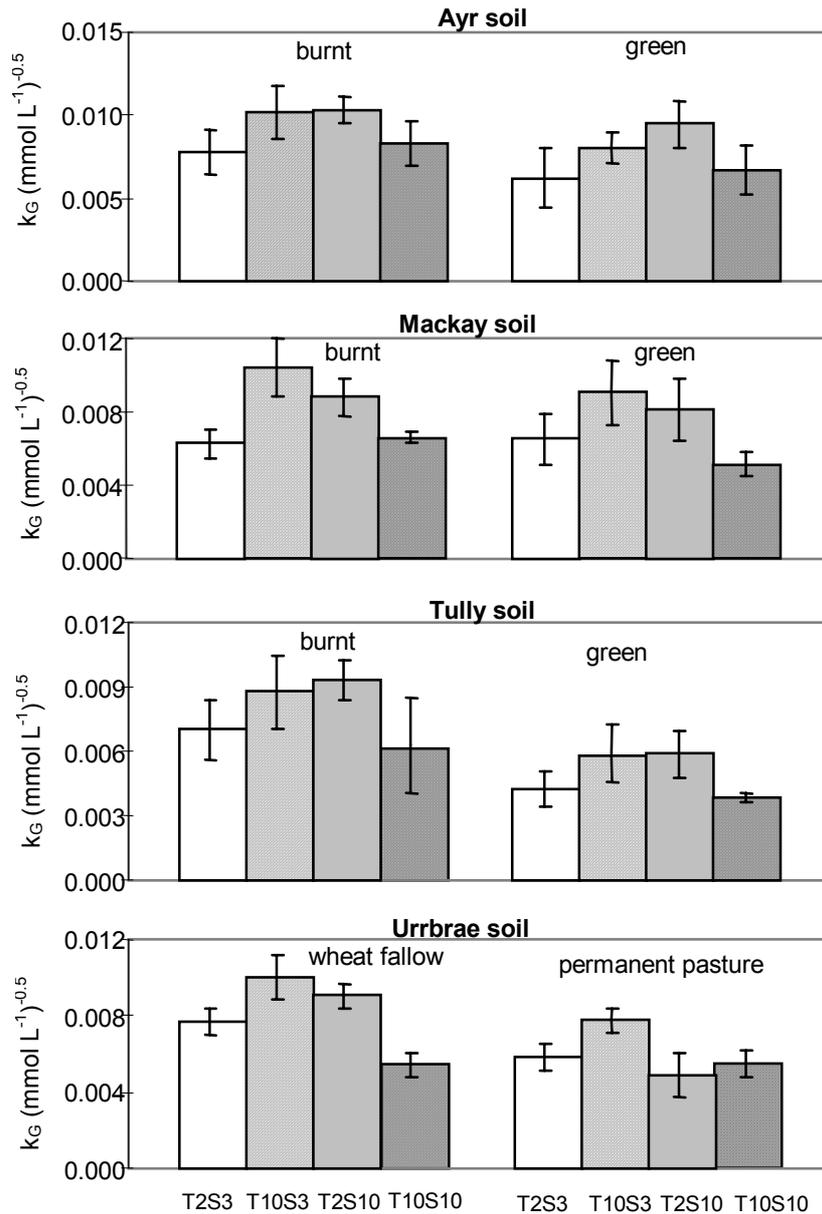


Figure 28 Effects of total cation concentration (TCC) and sodium adsorption ratio (SAR) on the value of k_G for different trash management practices in each soil. T2, T10 & S3, S10 denote TCC 2, 10 mmol(+)L⁻¹ and SAR 3, 10, respectively

6.7 Conclusions

Decomposition of sugarcane trash was unaffected by high levels of soil sodicity (ESP 18-27) in a light- and a medium-textured soil, and was reduced by 18% in a heavier-textured soil. We attribute the latter effect to impaired soil aeration due to the combination of high sodicity, high clay content, high pH, and relatively low organic matter content. Decomposition was unaffected by milder sodicity (ESP 8-12). On the basis of these findings, it is unlikely that soil sodicity will be an important limit to trash decomposition in sugarcane fields where viable crops can be grown.

Molasses and gypsum both increased the structural stability of sodic soils used in this study, but the effects depended to some extent on clay content. The effects of gypsum were more pronounced in the Burdekin soil (higher clay content), particularly in relation to reducing clay dispersion. On the other hand, molasses alone increased macroaggregation in the both soils regardless of soil texture. On the basis of this laboratory work, it is, therefore, likely that the addition of molasses alone or in combination with gypsum may be beneficial for overcoming problems with sodic soils, particularly those of higher clay content used for either sugarcane production or in regions where molasses is available relatively cheaply.

Molasses and lime improved the structural stability and reduced the ESP of both sodic soils and both show promise as ameliorants either alone or in combination. In our experiment, the rate of 10 t/ha molasses did not lower the pH of the soil sufficiently to dissolve the lime substantially. Further experiments are needed to determine the best combination rates of molasses and lime to overcome problems associated with sodicity.

The method used for measuring cation exchange selectivity, using minimal disturbance to soil aggregates, proved successful. Cation exchange selectivity was significantly influenced by trash management in all soils due to changes in the content and nature of soil organic matter. GCTB can significantly reduce the tendency of sugarcane-growing topsoils to become sodic or remain sodic compared to a burnt system.

Cane trash, other organic ameliorants such as molasses, and soil organic matter all have an important role to play in the management of sodic soils. All forms of organic additions examined improved the structural properties of sodic soils, and reduced sodicity or the tendency of soil to become or remain sodic. There were no major interactions between organic matter and Ca-containing ameliorants; they all had positive effects irrespective of each other.

7.0 EFFECT OF CALCIUM-BASED AMENDMENTS ON TRASH DECOMPOSITION

7.1 Introduction

Although the advantages of retaining sugarcane crop residues as a trash blanket have been well documented (Wood 1991; Thorburn *et al.* 1999), nutrient cycling in relation to GCTB is not fully understood. As part of a Cooperative Research Centre for Sustainable Sugar Production (CRC Sugar) subprogram, researchers have examined the return of nutrients to soil and the factors that affect the breakdown of organic matter associated with trash management systems (Thorburn *et al.* 2001; Robertson and Thorburn 2002). Despite the many advantages of nutrient cycling and release (Mitchell, and Larsen 2000; Robertson and Thorburn 2000; Klok and Kingston 2002) and soil ameliorative effects of trash retention (Moody 2002), advantages of enhanced decomposition, slow trash decomposition rates in some circumstances (Kingston *et al.* 1998), and reports of possibly acidification with trash retention (AD Noble, pers. comm.) have indicated that there is a need to further develop the trash retention system.

Although lime and gypsum are routinely applied to sugarcane soils to supply calcium (Calcino 1994; Kingston and Aitken 1996), and lime is used to ameliorate acidic conditions (Noble *et al.* 1997, Aitken *et al.* 1998), little is known about the effects of applying calcium-containing amendments to sugarcane trash, both in relation to trash decomposition and properties of the underlying soils.

This section reports on the results of an investigation aimed at determining the effects of applying calcium-based amendments on trash decomposition. The investigation consisted of two phases. Phase one dealt with the effect of various amendments on the rate of trash decomposition and the subsequent ameliorative effect on the underlying soil. In phase two, we investigated the nutrient release and uptake associated with this interaction.

7.2 Effect of various calcium amendments on trash decomposition and soil amelioration

7.2.1 Method and materials

This investigation consisted of four separate but interlinked experiments.

Experiment 1

This uncropped pot experiment, conducted in the Queensland Department of Natural Resources and Mines (NRM) glasshouse at Indooroopilly, was used to determine the effect of applying lime to trash.

Two acidic topsoils (hydrosol and kurosol) with differing chemical properties (Table 34) were used in a glasshouse experiment. Soil (50 mm deep), either unamended or amended with lime, was placed in trays (300 mm x 400 mm) and wet to field capacity. A 50-mm-thick blanket of cane trash was added to each tray. Lime application rates of 0, 3 and 6 t/ha were applied on the surface of the trash or incorporated into the soil prior to spreading the trash. An additional treatment was established involving unamended soil, trash addition and urea (160 kg N/ha) spread on the surface of the trash. Field moisture capacity was maintained by evenly spraying water onto the surface of the trash in each tray to predetermined tray masses.

Table 34 Initial chemical properties of soils used in Experiment 1

Soil	pH (water)	pH (CaCl ₂)	Org C (%)	Exch Ca	Exch K	ECEC	Al	Mn
				cmol(+)/kg				
Hydrosol	3.85	3.61	1.01	2.02	0.31	12.5	110.6	7.3
Kurosol	4.95	3.87	1.05	0.31	0.10	3.10	46.5	28.9

Trays were destructively sampled after 4 and 7 months. The undecomposed trash, the 0-10 mm soil/decomposed trash layer and the 10-50 mm soil layer were sampled separately. The extent of decomposition of trash was evaluated by measuring the dry mass of

remaining trash. The chemical characteristics of the residual trash and both soil layers were determined, as was the microbial biomass in the 0-10 mm soil layer (at 4 months).

Experiments 2 and 3

Two fully randomised pot trials were established, one at the NRM glasshouse at Indooroopilly (subtropical conditions) and the other in the glasshouse at CSR Sugar Macknade Mill at Ingham (in the moist tropics). The experiment conducted at Indooroopilly included five different soil types: a hydrosol from Coolum, a kurosol from Yandina, a humic podosol from Nambour, a dermosol from Ingham, and a vertosol from Ayr. The chemical and physical properties of these soils are presented in Table 35. The experiment at Macknade included a rudosol (river levee soil) and a tenosol (flood plain alluvial soil), both from the Herbert River sugarcane area, as well as the kurosol and dermosol soils used in the Indooroopilly experiment for cross referencing purposes (Table 35).

Table 35 Initial chemical properties of soils used in Experiments 2 and 3

Soil	Local soil identifier	pH water	Org C (%)	K	Ca	Mg	Na	Al	ECEC
				(cmol(+)/kg)					
Experiment 2									
Hydrosol	Garrett	4.5		0.63	6.34	3.59	1.52	4.34	16.42
Kurosol	McCord	4.7	1.0	0.18	0.36	0.38	0.06	2.31	3.29
Podosol	Cook	5.2	3.3	0.21	1.41	1.16	0.21	1.54	4.63
Vertosol	Barratta	7.0	1.4	0.26	11.78	6.51	0.61	-	19.16
Kandosol	Red loam	5.5	1.3	0.49	1.83	0.59	0.03	0.38	3.32
Experiment 3									
Kurosol	McCord	4.7	1.0	0.18	0.36	0.38	0.06	2.31	3.29
Kandosol	Red loam	5.5	1.3	0.49	1.83	0.59	0.03	0.38	3.32
Tenosol	River Bank	5.0	1.0	0.18	1.45	0.94	0.06	1.44	4.07
Dermosol	River Overflow	5.3	1.6	0.32	6.98	2.54	0.08	0.07	9.99

In both trials, soil (80 mm deep) was placed in square containers (160 mm x 160 mm) and partitioned into different depth intervals (0-10, 10-20, 20-50 and 50-88 mm below the soil surface) using plastic coarse-meshed grids. With the exception of the treatments without trash, effectively 40 t ha⁻¹ of air-dried trash was applied to the soil surface in each container. Various amendments were then spread on the surface of the trash (Table 36). As the experiment conducted at Indooroopilly was aimed at simulating weather conditions in the subtropics, temperatures in the glasshouse ranged from a minimum of 10°C to a maximum of 30°C. The pots were watered regularly at average rates of 300 mL twice weekly. In the Macknade trial, moist tropical conditions were simulated. Glasshouse temperatures ranged from a minimum of 15°C to a maximum of 40°C and the pots were watered regularly at average rates of 500 mL twice weekly.

The pots were destructively sampled after 7 months by retrieving the residual trash in each pot. The residual trash was determined by the combining the obviously undecomposed trash layer with the organic material remaining after the soil/trash interlayer was washed with deionised water to eliminate any soil particles and/or inorganic residues. This

residual trash was oven dried at 70°C and weighed. Soil pH (water) values were determined for each soil sample after they had been air-dried and ground to pass through a 2-mm screen.

Table 36 List of amendments included in the Indooroopilly (Experiment 2) and Macknade (Experiment 3) pot trials

Treatment	Amendment and equivalent rate*
1	Lime (calcium carbonate) at 3.0 t ha ⁻¹)
2	Gypsum (calcium sulphate) at 4.10 t ha ⁻¹
3	Calcium hydroxide at 2.22 t ha ⁻¹
4	Lime (1.5t ha ⁻¹) + gypsum (2.05 t ha ⁻¹)
5	Mill mud at 30t dry mud ha ⁻¹)
6	Potassium hydroxide 1.7 t ha ⁻¹)
7	Magnesium oxide at 1.2t ha ⁻¹
8	Urea at 160 kg N ha ⁻¹
9	Lime at 3.0 t ha ⁻¹ , No trash
10	No amendment, Trash only (40 t ha ⁻¹)
11**	No trash, no amendment

* Rates of amendments, other than that of millmud, were calculated to give Ca supply capacities and/or acid neutralising capabilities similar to 3 t/ha of calcium carbonate.

* Indooroopilly only

Experiment 4

This field trial was established at Macknade to evaluate the interaction between trash and lime/millmud under field conditions. Details are given in Table 37. Composite soil samples were collected from beneath the GCTB at CSR Technical Field Department: Block B14 after harvest of a first-ratoon crop, prior to the layout of the trial. The amount of trash originally present was determined by collecting trash from random areas of 1.5 m² between adjacent rows (row spacing of 1.5 m) along the full length of the block. At the same time, quadrats were established in each plot using coarse plastic mesh (covering a length of 3 m between rows) to facilitate measurement of trash decomposition. Treatments (Table 37) were applied on the surface of the trash. Immediately prior to harvest of the next crop, residual trash samples were collected from each quadrat. Composite soil samples were also collected from beneath each quadrat (0-5 cm and 5-10 cm depths). Organic C (0-50 mm soil depth), soil pH (0-50 mm and 50-100 mm soil depth) and exchangeable Al) were used to determined to assess the ameliorative effect of the various amendments applied to the surface of the trash blanket.

Table 37 Details of the field trial conducted at Macknade (Experiment 4)

Experiment 4		Macknade	
Design details:		Randomised block	
Site:		CSR Technical Field Department: Block 14	
Soil type:		River Bank (Tenosol)	
Variety:		Q164	Crop: After 1st Ratoon
Date established:		June 2000	Date harvested: July 2001
Mean soil values (pre-treatment)		Treatment applied	
Topsoil (0 – 200 mm)		Lime (2.5 tonnes ha ⁻¹)	
pH(water)	5.73	Lime (5 tonnes ha ⁻¹)	
Org C (%)	0.7	Millmud (200 tonnes ha ⁻¹) ex Victoria Mill	
BSES P (mg kg⁻¹)	123	Millmud/millash (200 tonnes ha ⁻¹) ex Macknade Mill	
Exch. K (cmol(+) kg⁻¹)	0.09	Control (unamended trash)	
Exch. Ca (cmol(+) kg⁻¹)	7.55		
Exch. Mg (cmol(+) kg⁻¹)	1.49		
Exch. Na (cmol(+) kg⁻¹)	0.15		
Extr. Al (cmol(+) kg⁻¹)	-		
Sulphate S (mg kg⁻¹)	5		
Clay (%)	12		
ECEC (cmol(+) kg⁻¹)	9.28		
Subsoil			
pH (water)	6.72		
Exch. K (cmol(+) kg⁻¹)	0.06		
Exch. Ca (cmol(+) kg⁻¹)	8.53		
Exch. Mg (cmol(+) kg⁻¹)	2.27		
Exch. Na (cmol(+) kg⁻¹)	0.18		
Extr. Al (cmol(+) kg⁻¹)	-		
ECEC (cmol(+) kg⁻¹)	11.04		

7.2.2 Results and discussion

Experiment 1

Samples taken at 4 months after establishment indicated that lime had a significant effect on trash decomposition, but the extent of decomposition was dependent on the method of lime application and soil type (Table 38), although no interactive effects were apparent (Figure 29). Spreading lime on the surface of the trash layer enhanced decomposition. This was attributed to possible increased solubility of organic matter and improved microbial activity with increased pH within the decomposing trash blanket. Although there was no significant treatment effect on microbial biomass with increased pH, it was highest in the surface soil (0-10 mm) where lime (6 t/ha) was spread onto the trash (data not shown here). Although urea significantly increased trash decomposition across soils, it was slightly less effective than lime (3 t/ha).

Table 38 Amounts of residual trash after 4 months in Experiment 1

Soil	Residual trash (t/ha)	Lime applied (t/ha)	Residual trash (t/ha)	Application	Residual trash (t/ha)
Hydrosol	20.45	0	20.57	Spread on trash	20.36
Kurosol	19.61	3	19.83	Incorporated in soil	19.70
		6	19.70		
Significance	**		**		**
Lsd(p<0.05)	0.23		0.28		0.23
Lsd(p<0.01)	0.31		0.38		0.31

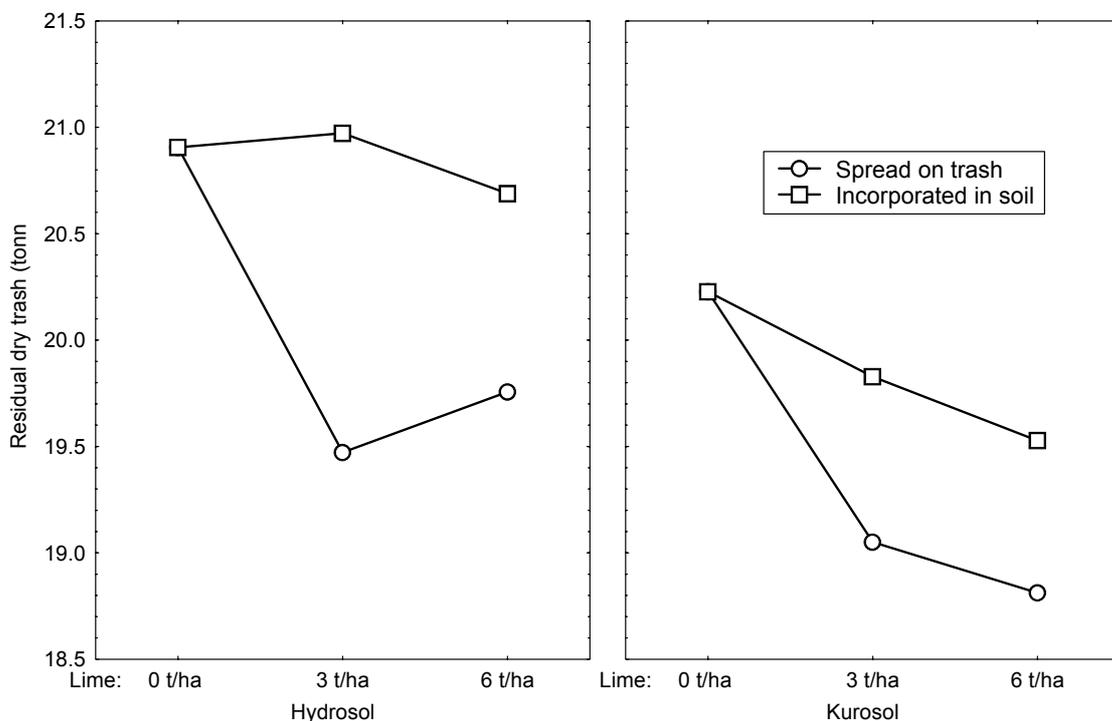


Figure 29 Residual trash as influenced by lime rate and method of application. Interactive effect (lime x method of application) was not statistically significant

The 7-month sampling showed that the differences in residual trash had markedly narrowed for both soils and no significant differences existed between treatments (Figure 30). The overall slow decomposition rate and loss in treatment differences have been attributed to the low volume of water applied. This resulted directly from the fact that the pots used had sealed bottoms and it was necessary to restrict water application to maintain the soil moisture content at field capacity.

As expected, incorporating lime into the soil increased soil pH in both the 0-10 mm (Figure 31) and 10-50 mm layers (Figure 32). Spreading lime onto the trash layer also resulted in marked increases in soil pH values in the surface soil (0-10 mm). Although there was evidence of small pH increases in the 10-50 mm soil layer from the application

of lime onto the surface of the trash after 7 months, these differences were not significant. However, the substantially lower exchangeable Al values associated with the 10-50 mm soil layer at 4 and 7 months (Figure 33) was evidence that some amelioration had occurred with the application of lime to the surface of the trash. These data offer encouraging evidence that the interaction between lime application and trash may have important implications for the amelioration of sub-surface soil acidity in minimum tillage systems.

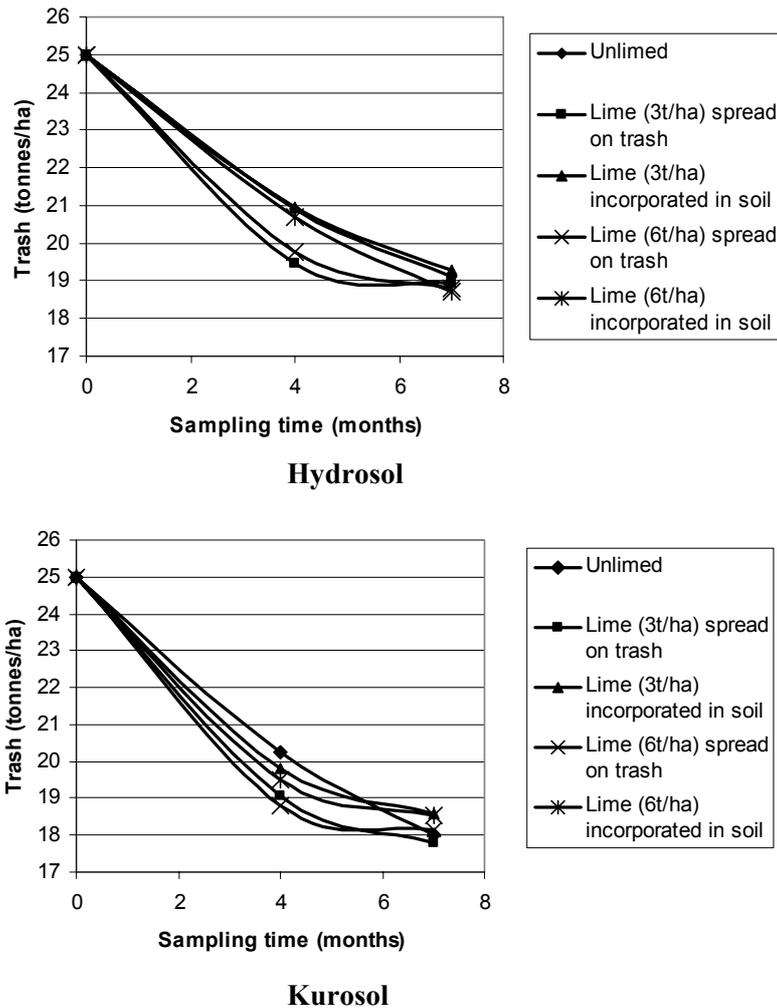


Figure 30 Trash decomposition as influenced by time and lime application

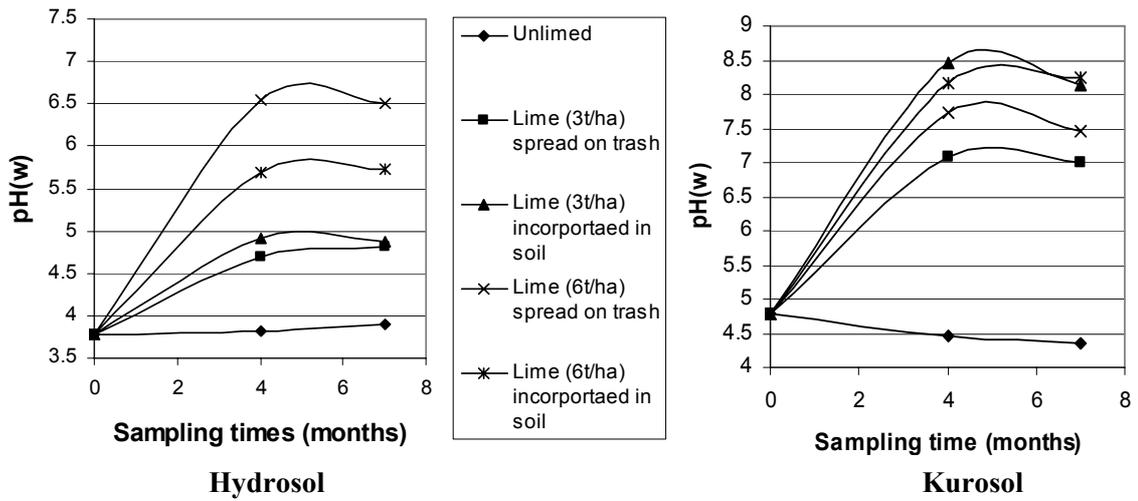


Figure 31 pH of the 0-10mm soil depth (immediately below the trash layer) as influenced by time and lime application

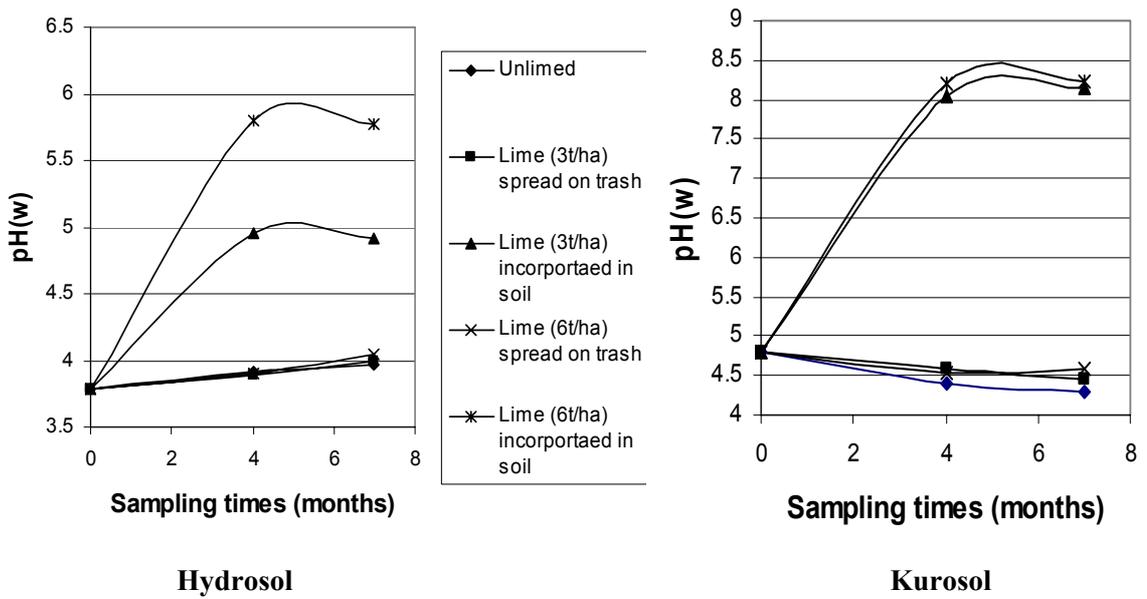


Figure 32 pH values of the 10-50 mm soil depth as influenced by time and lime application

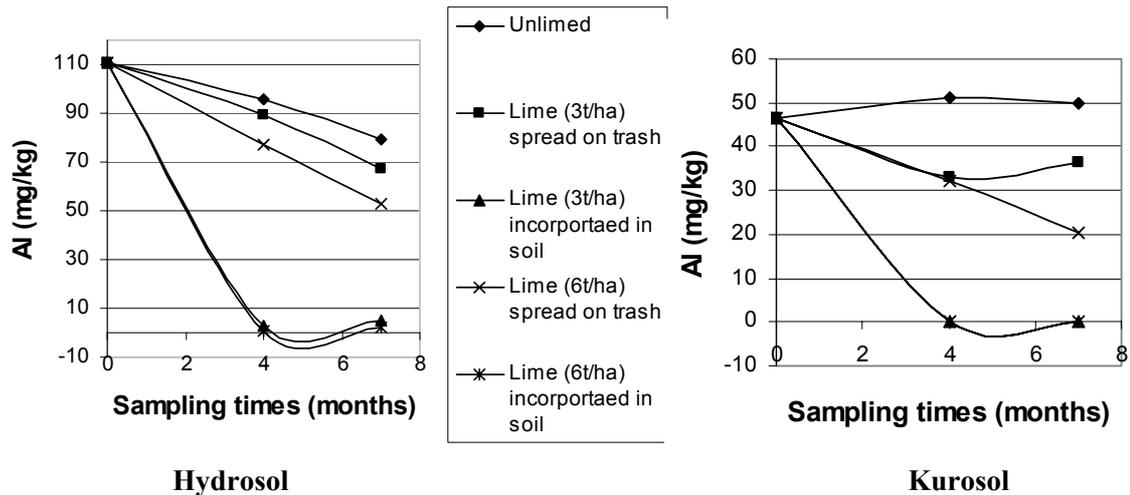


Figure 33 Exchangeable Al values of the 10-50 mm soil depth as influenced by lime application and time

Experiments 2 and 3

Trash decomposition in the Indooroopilly trial (Experiment 2) was affected by both soil type (Figure 34) and amendment applied (Figure 35). Residual trash (mass of remaining trash expressed as a percentage of the original mass of trash and amendment applied) indicated that although decomposition was similar over the 7-month period on the kurosol, podosol and kandosol, the amount of residual trash on the hydrosol was significantly lower than that on the vertosol. This difference could be related to the origin and pH of the two soils. The vertosol is from an area of relatively low rainfall (Ayr) and has a pH value of 7 (highest in the range of soils considered – Table 35). The hydrosol is a strongly acid soil from Coolum. It is found in an area characterised by moderately high rainfall and subject to periodic water-logged conditions. It is possible that anaerobic micro-organisms would naturally be associated with soils in such bottom-land positions. As the pots received about 600 mL of water each week, the interface between the trash and soil would have been moist for much of the 7-month period. Although the application of lime and a combination of lime and gypsum resulted in residual trash levels significantly lower ($P = 0.05$) than that of unamended trash, mill mud (applied at a rate of 30 t of dry material) resulted in residual trash levels markedly lower (about 30% units) than any of the other treatments (Figure 35). The application of KOH also resulted in residual trash levels significantly lower than the unamended trash and the majority of the other treatments. This may have been the result of alkaline hydrolysis of the organic matter.

In Experiment 3 at Macknade, the application of millmud again resulted in the most marked decomposition of the trash layer (Figure 36). In this case, however, application of the other amendments (including lime and the lime/gypsum mixture) did not give rise to significantly lower levels of residual trash. Trash decomposition was not affected by soil type (Figure 37).

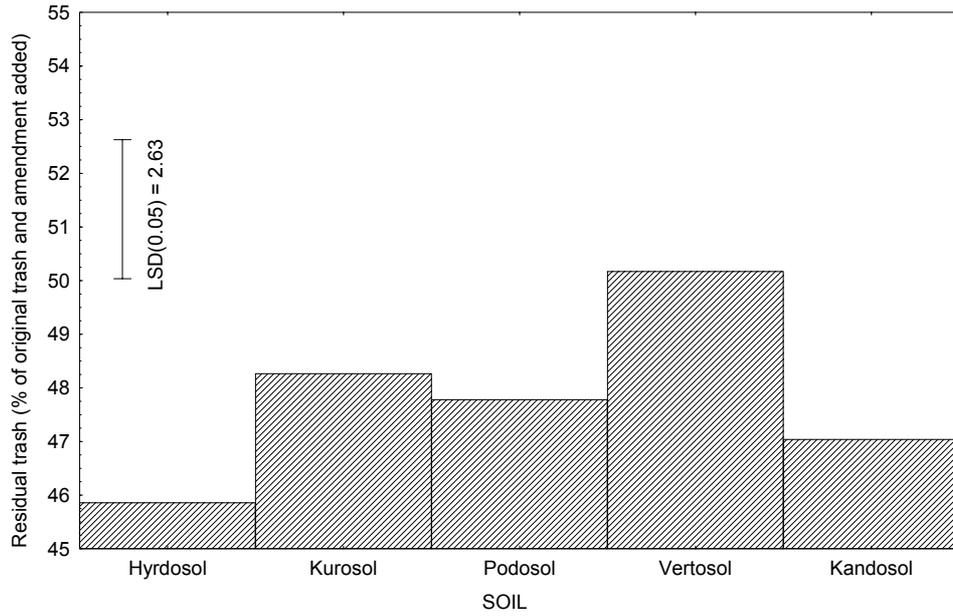


Figure 34 Residual trash (expressed as a percentage of the mass of original trash and amendment applied) associated with each of the soil types after a 7-month period in Experiment 2

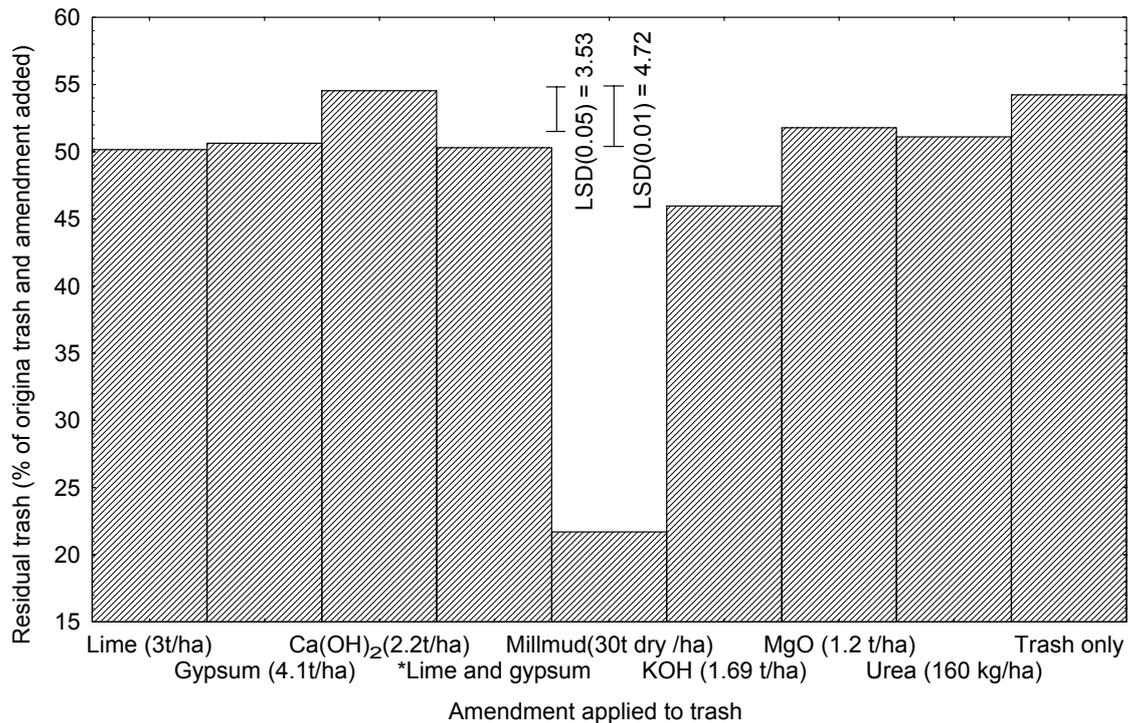


Figure 35 Residual trash (expressed as a percentage of the mass of original trash and amendment applied) as influenced by the type of amendment applied to the trash blanket in Experiment 2

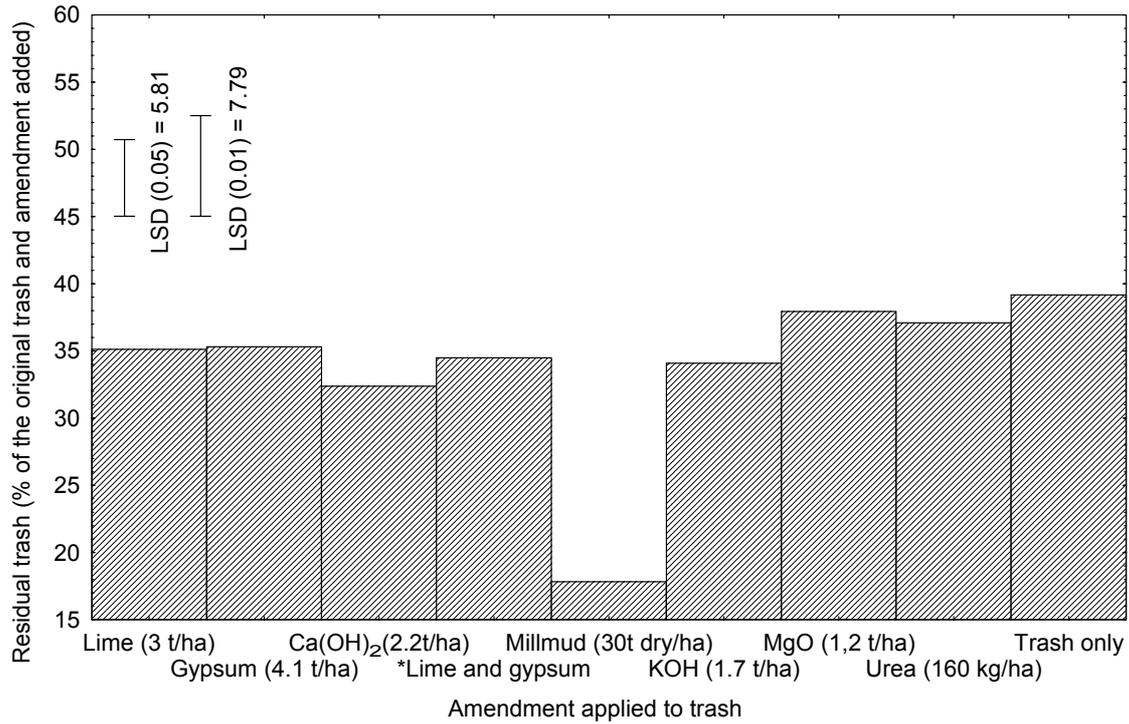


Figure 36 Residual trash (expressed as a percentage of the mass of original trash and amendment applied) as influenced by the type of amendment applied to the trash blanket in Experiment 3

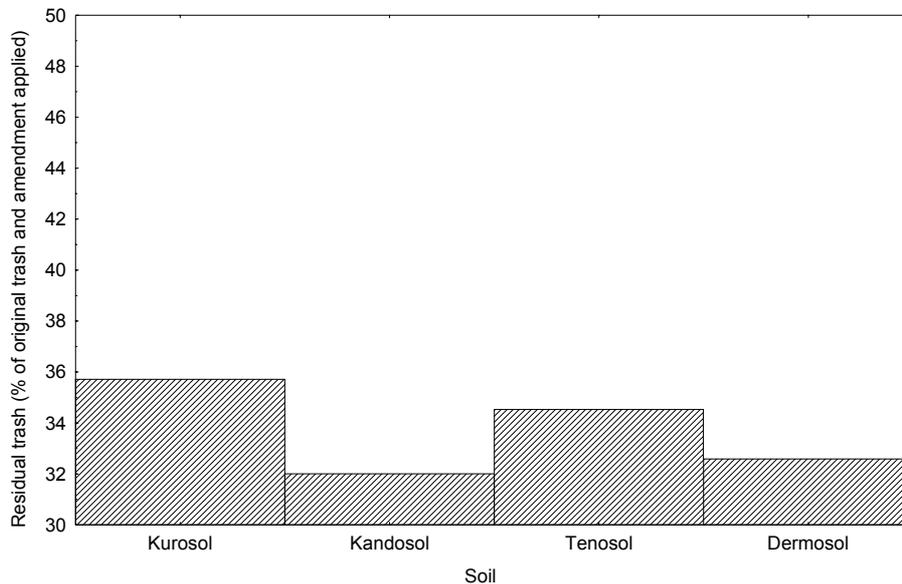


Figure 37 Residual trash (expressed as a percentage of the mass of original trash and amendment applied) associated with each of the soil types after a 7-month period in Experiment 3. Differences were not statistically significant

When the residual trash data relating to the soils common to the two experiments (kurosol and kandosol) were pooled, it was found that similar trends to the applied amendments

were apparent (Table 39). Although the amount (dry mass) of residual trash on the kandosol was significantly lower than that on the kurosol (probably due to better aeration), the most marked difference in residual trash was due to the different environments, with the mean amount of residual trash in the moist tropical environment significantly lower ($P < 0.01$) than the mean amount of trash remaining in the simulated subtropical environment. When considering the interaction between the simulated environment and amendment applied (Figure 38), we found that millmud was the only amendment that did not result in a significant difference in residual trash between the subtropical and moist tropical conditions. This means that millmud is effective in enhancing trash decomposition across the two simulated environments. This result is also well illustrated when relative residual trash (the mass of residual trash associated with each amendment expressed as a percentage of the mass of unamended trash remaining after the 7-month period) was plotted against the type of amendment applied in the subtropical (Figure 39) and moist tropical (Figure 40) environments. In the simulated subtropical conditions (Figure 39) the application of lime, KOH and millmud resulted in significantly lower relative residual trash values. In contrast, only the millmud application resulted in a significantly lower relative residual trash value in the simulated moist tropical environment (Figure 40). It would appear that the more severe conditions (larger amounts of water applied and/or higher temperatures) in the tropical environment ultimately negated any enhanced decomposition effect due to lime (and KOH) application as observed in the subtropical environment. This supports the results of Experiment 1, which showed that the significant differences in residual trash values due to lime application (at 4 months) was eliminated after 7 months.

Table 39 Trash (expressed as a percentage of the original trash and amendment applied) remaining above the two common soils (Kurosol and Kandosol) included in the simulated subtropical and moist tropical environments after 7 months

Simulated environment	Residual trash (%)	Soil	Residual trash (%)	Amendment added to trash	Residual trash (%)
Subtropics	47.65	Kurosol	41.99	Lime (3 t/ha)	40.19
Moist tropics	33.86	Kandosol	39.52	Gypsum (4.1 t/ha)	43.38
				Ca(OH) ₂ (2.2 t/ha)	45.53
				Lime (1.5 t/ha) + gypsum (2.05 t/ha)	43.13
				Millmud (30 dry t/ha)	20.95
				KOH (1.7 t/ha)	37.61
				MgO (1.2 t/ha)	45.10
				Urea (160 kg/ha)	43.81
				Trash only	47.10
Significance	**		*		**
LSD(p=0.05)	2.27		2.27		0.23
LSD(p=0.01)	3.05				0.31
Interaction				Significance	
Simulated environment x amendment				*	
Other interactions				NS	

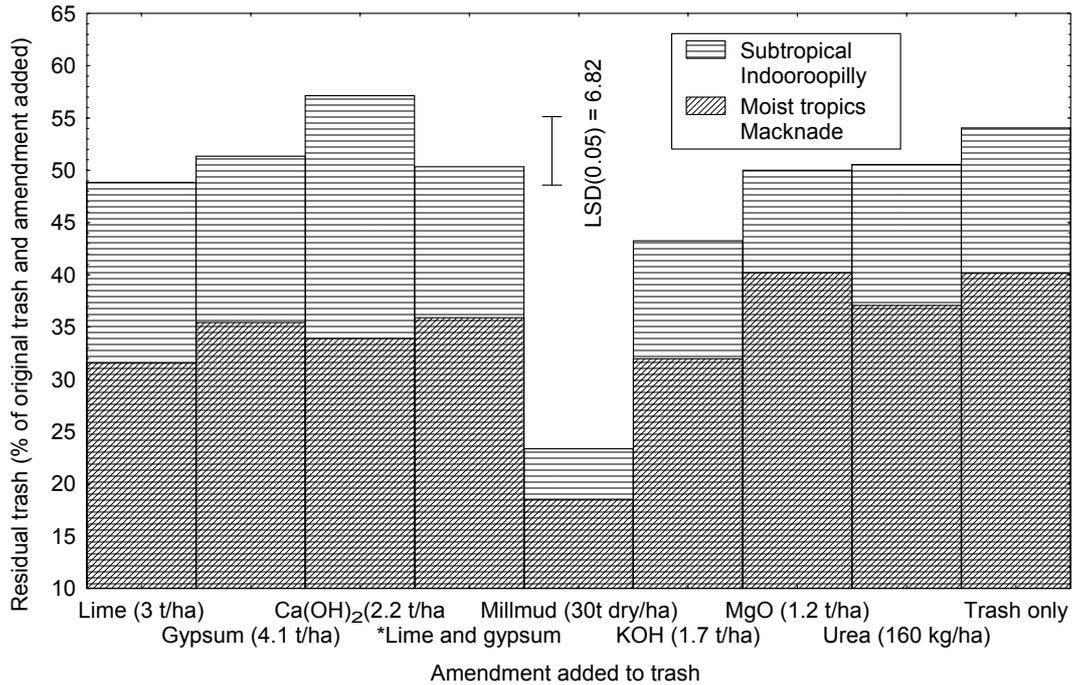


Figure 38 Residual trash as influenced by environment (subtropical versus moist tropical) and amendment applied

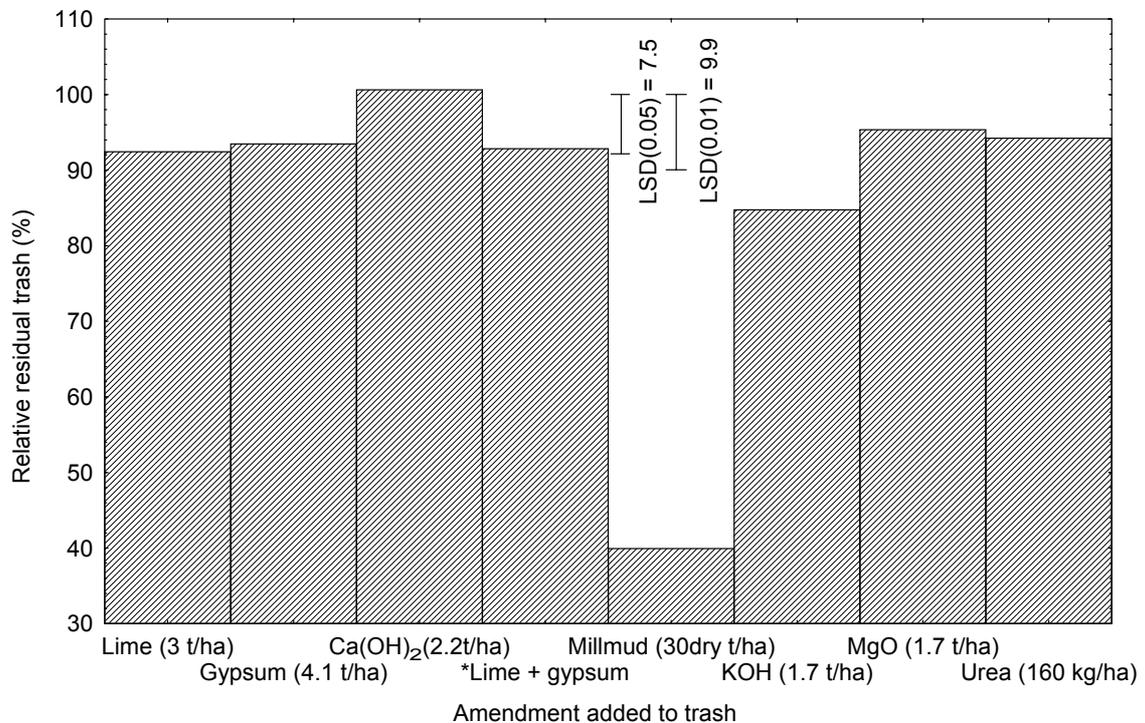


Figure 39 Relative residual trash values (expressed as a percentage of the unamended trash remaining after the 7-month period) in the simulated subtropical environment

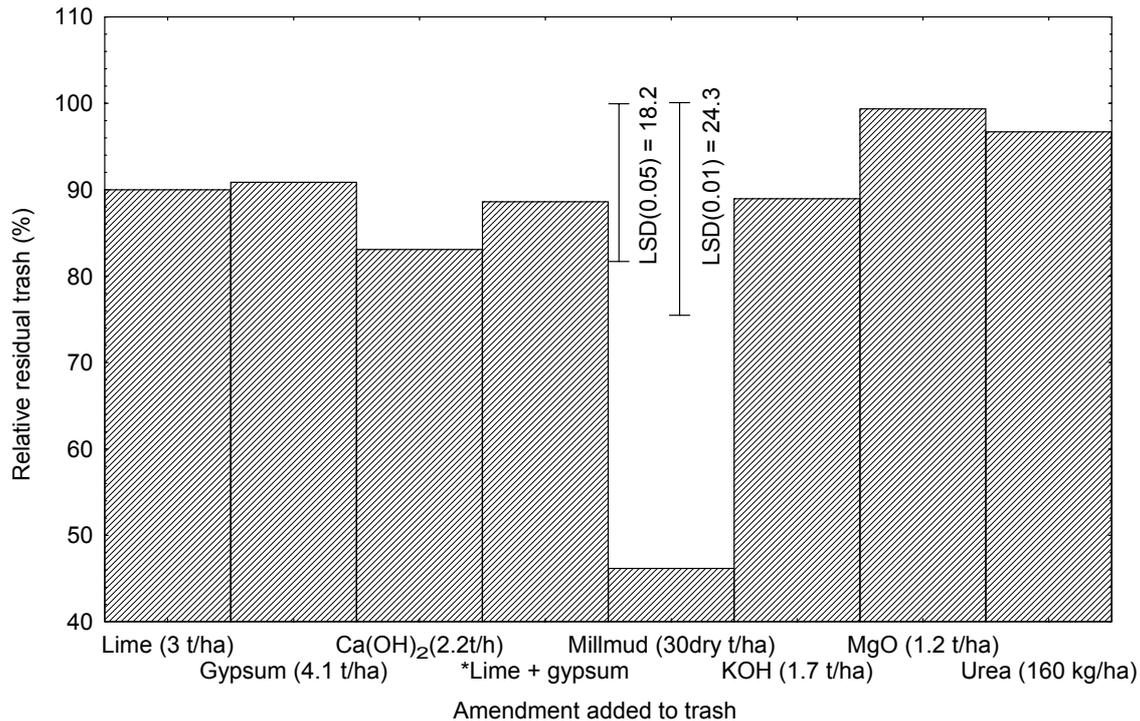


Figure 40 Relative residual trash values (expressed as a percentage of the unamended trash remaining after the 7-month period) in the simulated moist tropical environment

In assessing the ameliorative effect of applying an amendment to the trash blanket, only selected treatments were considered: lime on trash, millmud on trash, surface applied lime (no trash), trash (no amendment), and no trash or amendment (Experiment 3 only). In Experiment 2 (subtropical environment) soil type, treatment and depth below the trash layer had a significant interactive effect on soil pH (Figure 41). As expected, the surface applied lime increased soil pH across the five soil types. This was most pronounced in the surface layers and generally declined with depth. The presence of a trash blanket (in the absence of any amendments) had variable effects on soil pH. While surface applied trash appeared to increase the pH of the kurosol, humic podosol and kandosol, the effect was less marked in the hydrosol possibly due to a higher buffering capacity. In contrast the presence of trash resulted in a decrease in soil pH in the vertosol. The addition of lime to the surface of the trash layer also resulted in variable effects. In the hydrosol, podosol and kandosol, the effect on soil pH was either similar or less than that of surface applied lime (no trash). However, in kurosol, lime application to the trash blanket resulted in higher pH values in the sub-surface layers than when lime was applied in the absence of trash, thus supporting earlier findings.

In experiment 3 (moist tropical environment), pH trends were similar to those that occurred in Experiment 2 in the acid soils, particularly in the kurosol and kandosol, which were common to both experiments (Figure 42). The general increase in soil pH with trash retention was less marked in the dermosol than in the other soils. Once again, this is thought to be due to a higher buffering capacity. Lime application to trash on the kurosol again resulted in higher pH values in the subsurface layers than when lime was applied in

the absence of trash. This has important implications for certain acid soils where amelioration of sub-surface layers can be enhanced by this interaction of lime and trash, possibly due to more soluble organo-calcium complexes.

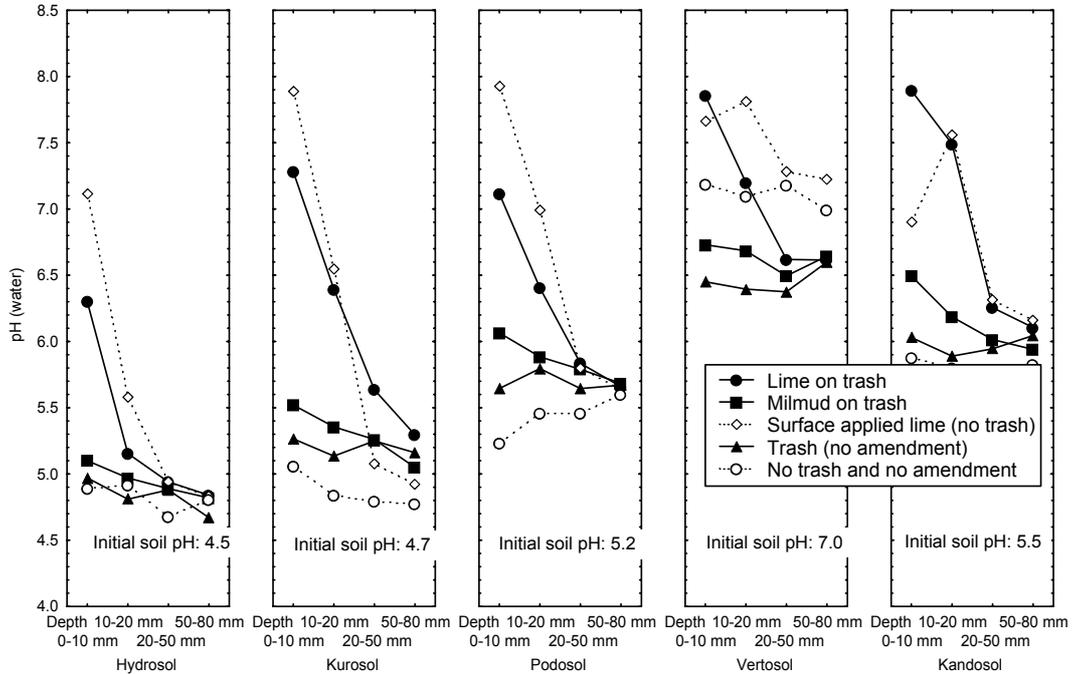


Figure 41 Interactive effect of soil type, treatment and depth on soil pH in simulated subtropical conditions. $LSD(0.01) = 0.46$

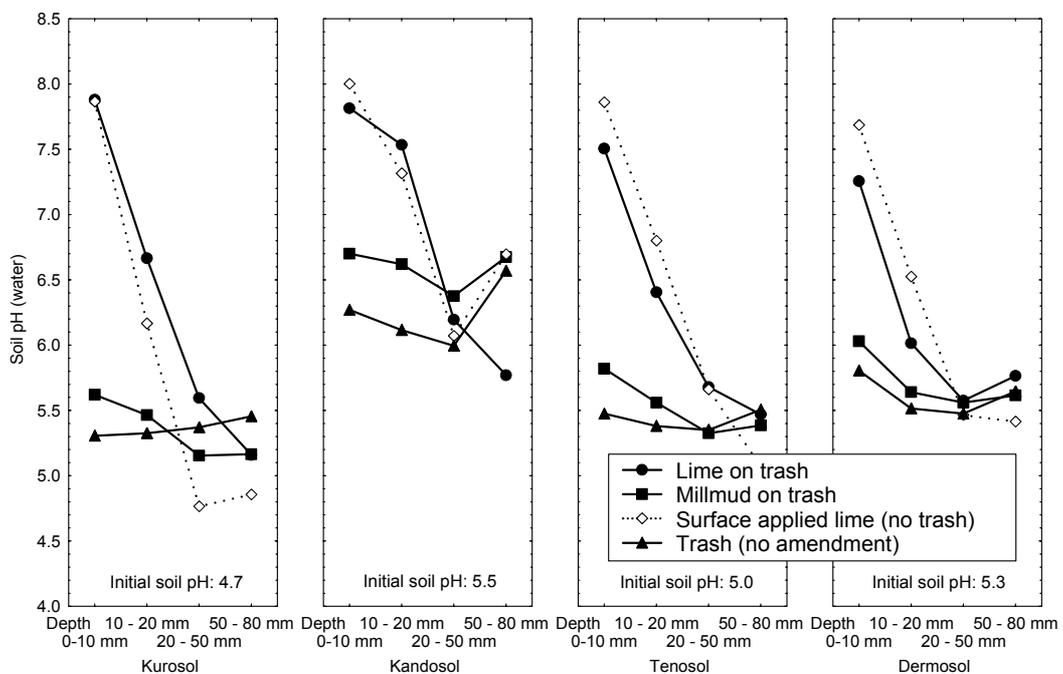


Figure 42 Interactive effect of soil type, treatment and depth on soil pH in simulated tropical conditions. $LSD(0.05) = 0.77$; $LSD(0.01) = 0.84$

Experiment 4

In the field trial at Macknade, all but the lime (applied at 2.5 t/ha) resulted in significantly less trash than the control (trash with no amendment added) after 12 months (Figure 43). Millmud (200 t wet material /ha (equivalent of 60 dry t/ha)), millmud/ash combination (200 t wet matter/ha (equivalent of 60 dry t/ha) and lime (5 t/ha) were equally effective in their ability to decompose trash.

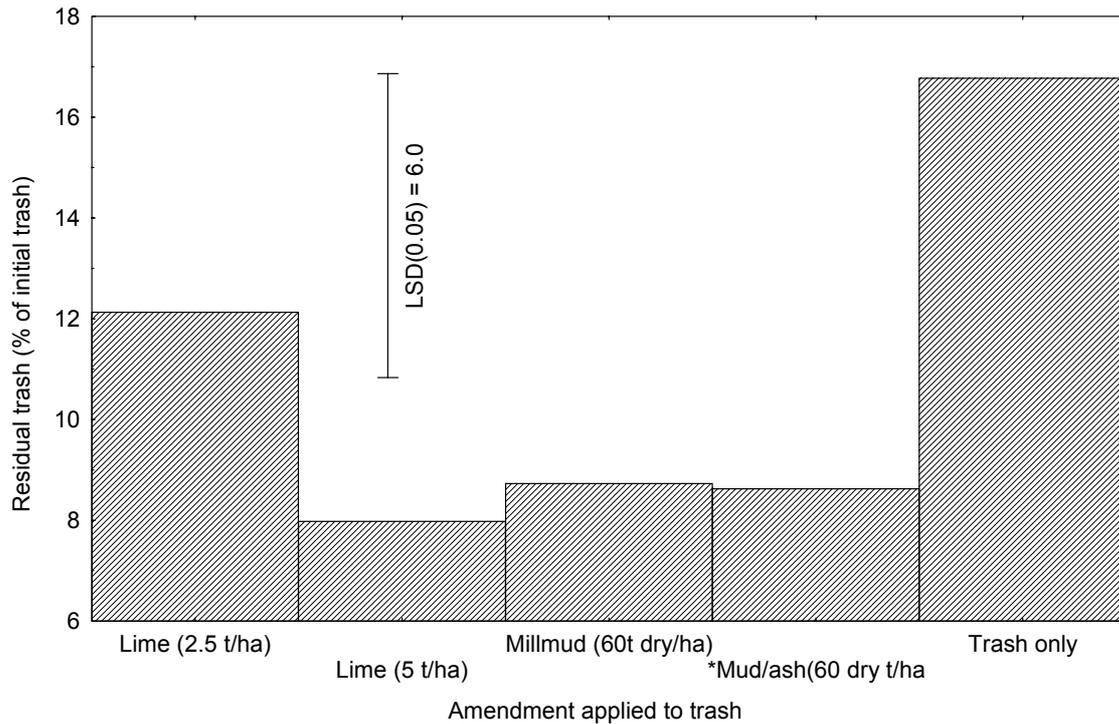


Figure 43 Residual trash (% of initial trash present) associated with each of the amendments applied to the surface of the trash blanket in the field trail at Macknade (Experiment 4)

A highly significant ($P < 0.01$) increase in soil organic C occurred in the soil layer 0-50 mm below the soil-trash interface when millmud and the millmud/mill ash mixture were applied (Figure 44). These increases were presumably, to a large extent, the result of the substantial amounts of organic matter added, rather than due to direct inputs from the trash, despite enhanced decomposition (Figure 43). Although lime applied at 5 t/ha resulted in a significantly lower amount of residual trash, no increase in soil organic C was noted (Figure 44).

Although the initial pH value and exch. Ca value of the soil indicated that amelioration of acidic conditions was not required, soil pH values were affected by the application of the amendments to the trash blanket, with lime elevating the mean soil pH to 6.23 and 6.58 (over the full depth interval of 0-100 mm) when lime was applied at rates of 2.5 and 5 t/ha, respectively (Table 40). Although the millmud/mill ash mixture resulted in the largest increased in soil pH to 6.89, no shift in soil pH occurred with the millmud

application. The absence of change in soil pH was also apparent when the interactive effect of amendment applied to the trash blanket and depth below the trash-soil interface was considered (Figure 45). This is probably due to the increased buffer capacity of the soil due to increases in organic C (Figure 17). This soil inherently has a relatively low buffer capacity with a clay content of 12% and org C = 0.7% (Table 37). In general millmud/mill ash mixtures are generally lower in organic C than plain millmud and have added abilities to increase soil pH values due to the nature of the ash material. The absence of a change in soil pH when millmud was applied to trash has important implications for certain soils, particularly where enhanced trash decomposition is desirable without further elevation of soil pH above 5.5.

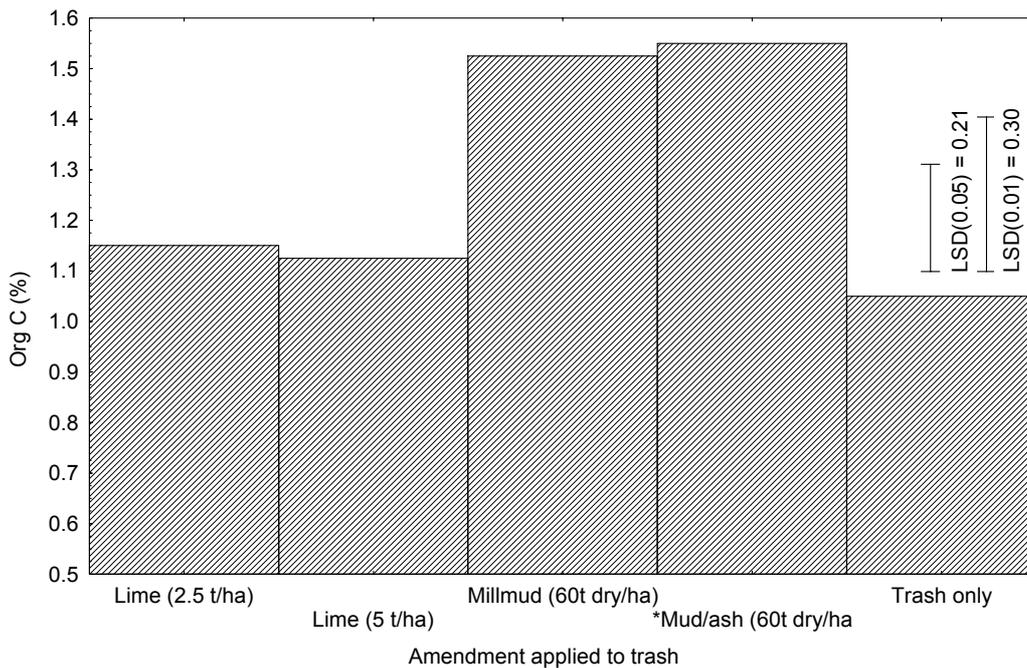


Figure 44 Organic C in the 0-50 mm soil depth immediately below the trash layer in the field trial at Macknade

Table 40 Soil pH values as affected by amendments applied to the surface of trash. Two soil depth classes were considered (0-50 mm and 50-100 mm below soil trash interface)

Soil depth (mm)	Soil pH(water)	Amendment added to trash	Soil pH(water)
0-50	6.16	Lime (2.5 t/ha)	6.23
50-100	5.80	Lime (5 t/ha)	6.58
		Millmud (60 dry t/ha)	5.60
		Millmud/mill ash (60 dry t/ha)	6.89
		Trash only	5.59
Significance	**		**
LSD (< 0.05)	0.11		0.18
LSD(<0.01)	0.15		0.24
Interaction			
Significance	**	LSD(< 0.05) = 0.26, LSD(< 0.01) = 0.34	

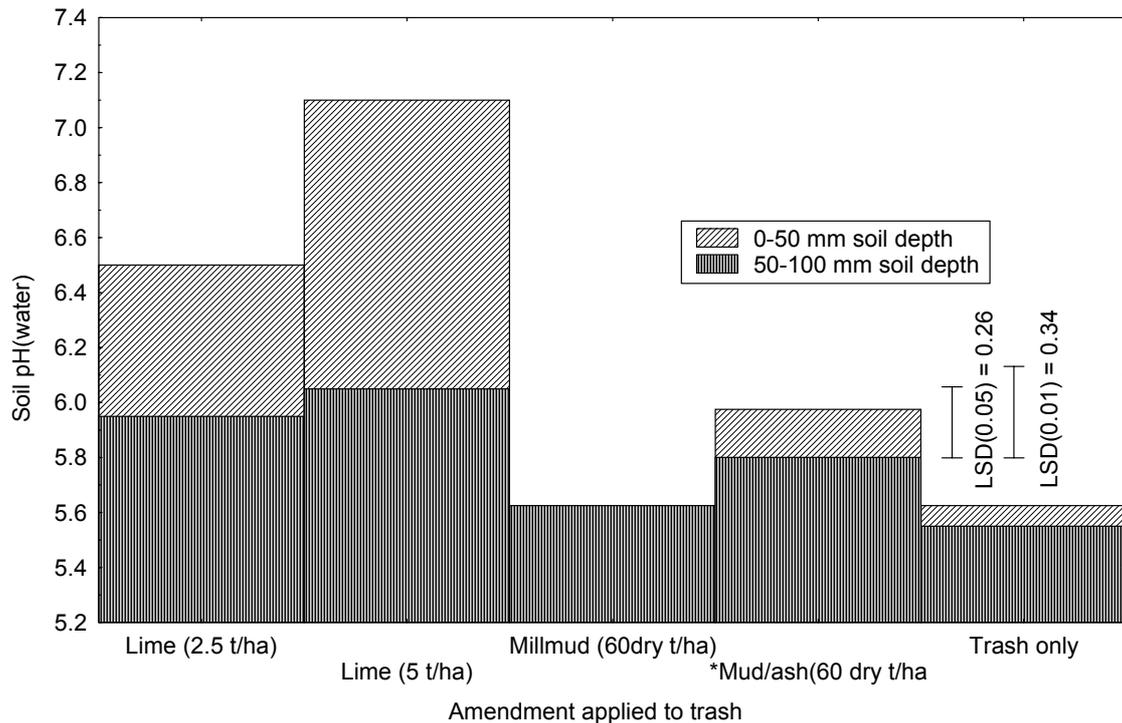


Figure 45 pH of the soil below the trash blanket as influenced by the application of various amendments

Despite already low exchangeable Al values in the soil prior to the application of various amendments to the trash (Table 37), lime treatments reduced the values to zero (Table 41), with millmud and millmud/mill ash mixture delivering an intermediate result.

Table 41 Exchangeable Al values of soil as affected by amendments applied to the surface of trash. Two soil depth classes were considered (0-5 mm and 5-10 mm below soil trash interface)

Soil depth (mm)	Exch. Al (cmol(+)/kg)	Amendment added to trash	Exch. Al (cmol(+)/kg)
0-5	0.035	Lime (2.5 t/ha)	0.000
5-10	0.024	Lime (5 t/ha)	0.000
		Millmud (60 dry t/ha)	0.044
		Millmud/mill ash (60 dry t/ha)	0.033
		Trash only	0.071
Significance	*	Significance	*
LSD (< 0.05)	0.011	LSD (< 0.05)	0.011
		LSD (< 0.01)	0.023
Interaction			
Significance	NS		

7.3 Combined effect of calcium-based amendments and trash retention on nutrient availability

This section of the investigation was made up of two separate but linked pot experiments. Both were aimed at determining whether the application of calcium-based amendments onto trash would deliver benefits in terms of nutrient availability to sugarcane. One trial (Experiment 5) was conducted at Macknade and included soils found in the Herbert River district. The other trial was conducted in Bundaberg and included three local (Bundaberg) soil and one soil from the Herbert region to enable cross-referencing.

Table 42 Selected chemical properties and physical attributes associated with the soils used in the pot experiments at Macknade (Experiment 5) and Bundaberg (Experiment 6)

Exp No	Soil type	pH (water)	Org C (%)	BSES P (mg/kg)	BSES Si (mg/kg)	Nitric K (cmol(+)/kg)	Cu (mg/kg)	Zn (mg/kg)
5	River sand	5.1	0.4	137	420	1.46	0.8	2.1
	Red loam	5.6	0.9	38	140	1.79	0.2	0.6
	River Overflow	5.3	1.6	102	1200	3.25	1.2	3.0
	Fine Black Sand	5.5	2.6	61	510	1.08	0.5	0.8
6	Woongarra	6.3	1.1	65	540	0.73	4.7	8.8
	River sand	5.2	0.4	176	390	1.68	0.7	2.2
	Oakwood	5.8	1.2	10	250	0.66	0.6	1.0
	Keppock	5.4	0.8	39	115	0.42	1.0	1.5
Accepted critical value*				40	70		0.2	0.6

* Calcino, 1994; Wood *et al.* 2003

Exp No	Soil type	Exchangeable actions					ECEC	Textural class
		K	Ca	Mg	Na	Al		
		(cmol(+)/kg)						
5	River sand	0.13	1.27	0.52	0.07	0.48	2.47	Loamy sand
	Red loam	0.19	3.04	0.44	0.02	0.13	3.82	Sandy clay loam
	River Overflow	0.32	6.98	2.54	0.08	0.07	7.69	Silty clay loam
	Fine Black Sand	0.10	3.27	0.39	0.09	0.23	4.08	Loamy sand
6	Woongarra	0.25	4.21	3.32	0.18	-	8.69	Clay loam
	River sand	0.13	1.29	0.51	0.08	0.50	2.51	Loamy sand
	Oakwood	0.38	3.56	2.40	0.09	0.07	6.50	Clay loam
	Keppock	0.30	3.86	1.07	0.16	0.07	5.46	Loam
Accepted critical value*		0.24	1.2	0.25				

* Calcino, 1994; Wood *et al.*, 2003

7.3.1 Methods and materials

Experiments 5 and 6

Sufficient quantities of soils ranging widely in chemical and physical properties (Table 42) were collected from the Herbert and Bundaberg districts, for two fully randomised pot experiments (Table 43). These soils (mixed with basal nutrient applications – Table 43) were placed in 0.25 m diameter freely draining plastic pots. Three previously germinated single-eyed setts of sugarcane (cultivars Q158 at Macknade, and Q141 and Q155 at

Bundaberg) were planted in each pot. Air-dried trash, cut to 50 mm lengths, was applied to each pot (equivalent of 15 t/ha). The amendments (Table 43) were spread on the surface of the trash layer in each pot. The chemical analyses of the millmud and mill ash are shown in Table 44. The pots were placed on exterior seedling benches and watered regularly with automatic overhead water sprinkling systems. The sugarcane plants were fully harvested at 6 months of age. The harvested plants were weighed and composite samples formed by combining the plant material from replicates 1 and 3, and 2 and 4. These samples were shredded, dried and re-weighed. The dried biomass was sent to a laboratory for chemical analysis.

Table 43 Details of pot experiments conducted at Macknade and Bundaberg

Pot experiment	5	6
Type	Soil x amendment Fully randomised	Soil x amendment Fully randomised
Soils	River Sand (Rudosol) Red Loam (Kandosol) River Overflow (Dermosol) Fine Black Sand (Hydrosol)	Woongarra (Ferrosol) River Sand (Rudosol) ex Macknade Oakwood (Kandosol) Kepnock (Yellow Dermosol)
Variety	Q158	Q155
Amendment applied to trash	Trash only Millmud (80 wet t/ha) Millash (80 wet t/ha) Lime (5 t/ha)	
Basal nutrient applications	100 kg N/ha 50 kg K/ha Trace element mix	
Replicates	4 (64 pots)	4 (64 pots)
Planting date	October 2000	Feb 2001
Harvest date	December 2000	June 2001
Age of cane at harvest	6 months	6 months

Table 44 Chemical analysis of amendments (dried) used in the pot experiments at Macknade and Bundaberg

Exp No	Amendment	Org C	N	P	K	Ca	Mg	S
		(%)						
5	Millmud	29.8	2.34	1.03	0.34	1.68	0.40	0.21
	Mill ash	31.1	0.61	0.38	0.84	0.72	0.40	0.22

7.3.2 Results and discussion

The mean dry-matter yield values associated with each of the soil types included in Experiment 5 (Figure 46) reflected the perceived potential of the soils to produce sugarcane crops. While the dermosol (River Overflow) is considered one of the most productive and fertile soils found in the Herbert River district (Wood *et al.* 2003), the other three are thought of as being moderately productive.

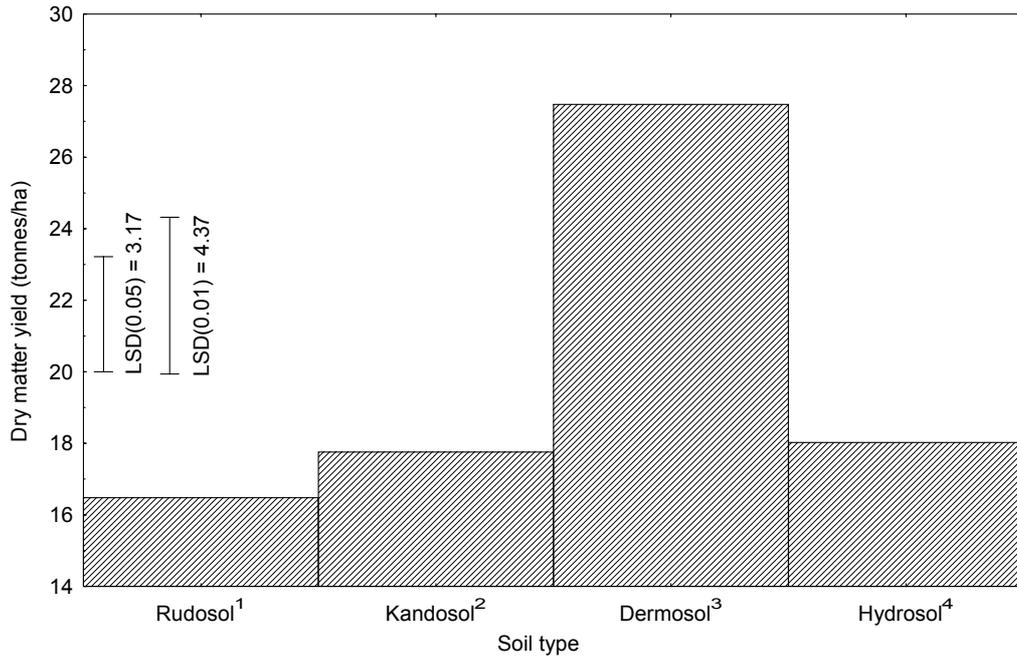


Figure 46 Mean dry matter yield values associated with each of the soil types included in Experiment 5

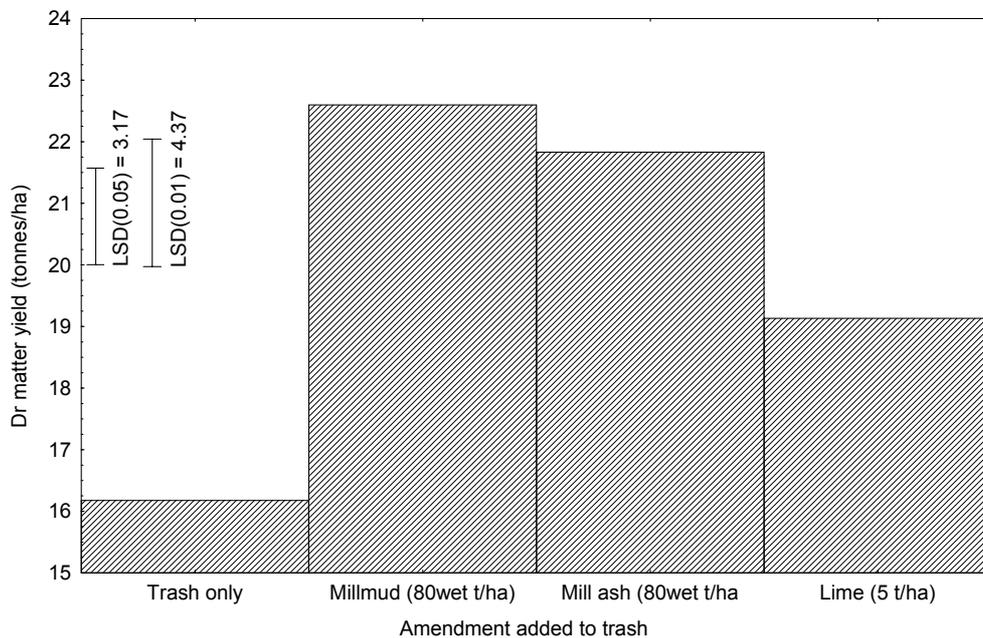


Figure 47 Mean dry matter yield values associated with each of the amendments applied to the trash in Experiment 5

The mean dry-matter yield values associated with the amendments applied to the trash in Experiment 5 (Figure 47) indicated that the largest response occurred due to the application of millmud (80 wet t/ha), but that the application of mill ash (80 wet t/ha) and

lime (5 t/ha) also resulted in significantly higher yields than the trash-only treatment. Although, both the mill mud and mill ash could have directly and indirectly improved the moisture status of the soil (compared with the trash only treatments), this would not have been the case when lime was added. In terms of the soil test values (Table 42) responses to applied nutrients other than N and K were unlikely (Wood *et al.* 2003), despite the large amounts of nutrients present in the millmud and mill ash. The responses to the amendments applied to the trash layer were therefore attributed, at least in part, to the enhanced decomposition of the trash and presumably the release of N from increased organic carbon. This effect occurred across all four soil types within the experiment, with the interactive effects (soil x amendments applied) being not statistically significant.

The amount of nutrients (N, P and K) removed by the harvested biomass expressed as kg/ha (Figure 48) were used to assess the nutrient availability resulting from the interaction between trash and the amendments applied. When considering the individual soils, the dermosol, as expected, supplied the largest amount of nutrients (257 kg N/ha, 36 kg P/ha and 237 kg K/ha). The rudosol supplied the smallest mean amounts of N and K, 122 kg N/ha and 124 kg K/ha, respectively, and the hydrosol supplied the least amount of P (17 kg/ha). In terms of amendments applied to the trash, the millmud and mill ash treatments resulted in significantly higher amounts of N, P and K removed by the harvested biomass (Figure 48). These elevated values could either have been the result of the nutrients within the amendments themselves, or due to enhanced decomposition of the trash, or a combination of both of these effects. Although the mean amounts of N and K removed by the harvested biomass when lime was applied to the trash were larger than that associated with the trash only treatment, these differences were not statistically significant. Phosphorus was the only nutrient to be influenced by a significant interactive effect between soil type and amendments applied (Figure 49). In the rudosol, kandosol and dermosol, significantly more P was removed by the harvested biomass when millmud and mill ash were applied to the trash than with unamended trash. In contrast, P removal on the hydrosol remained almost constant across all the trash/ amendment treatments. Possible reasons for this include the strongly P-sorbing character of this soil (Wood *et al.* 2003).

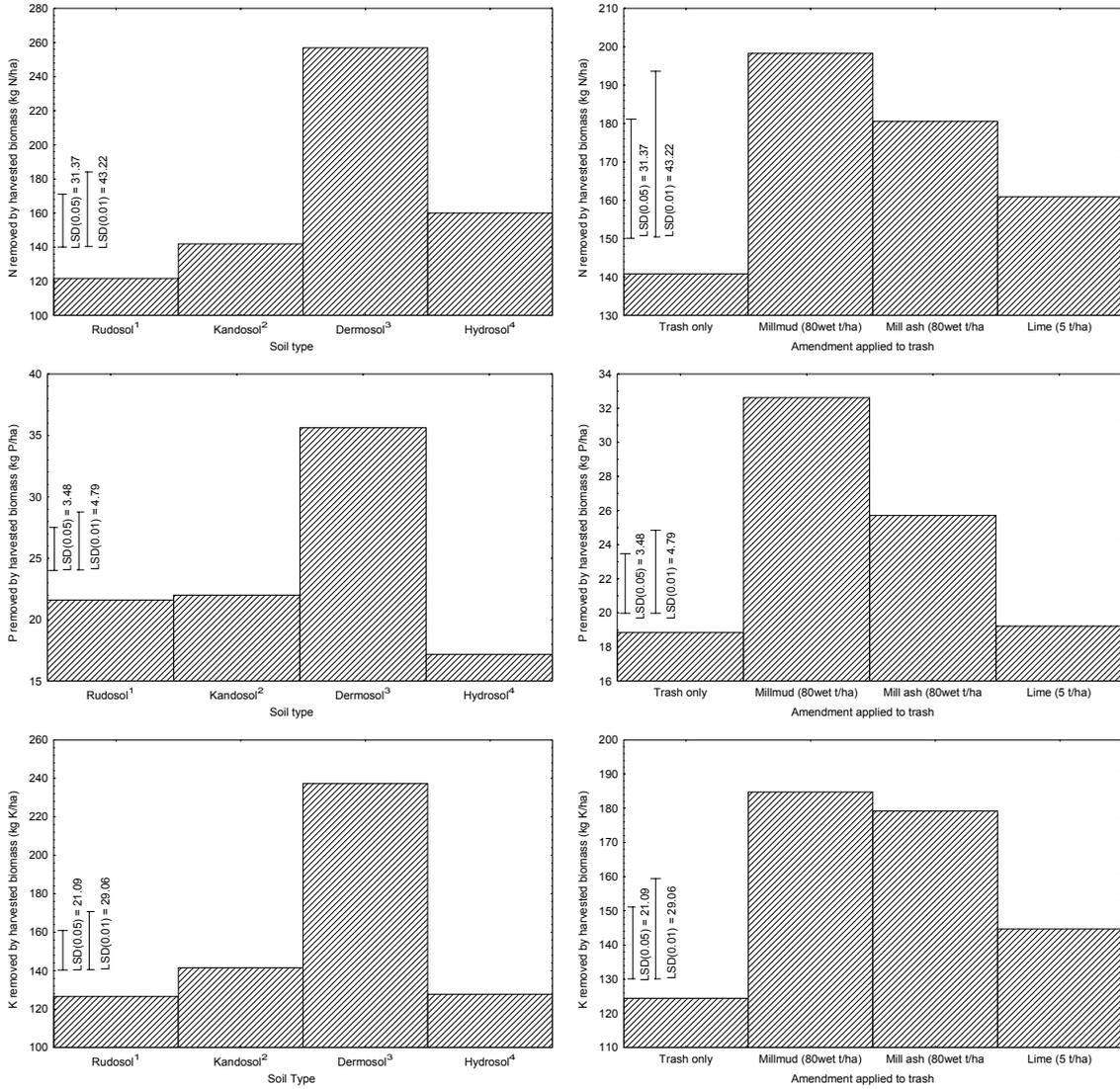


Figure 48 Nutrients (N, P and K) removed plotted against soil type and amendments applied to the trash in Experiment 5

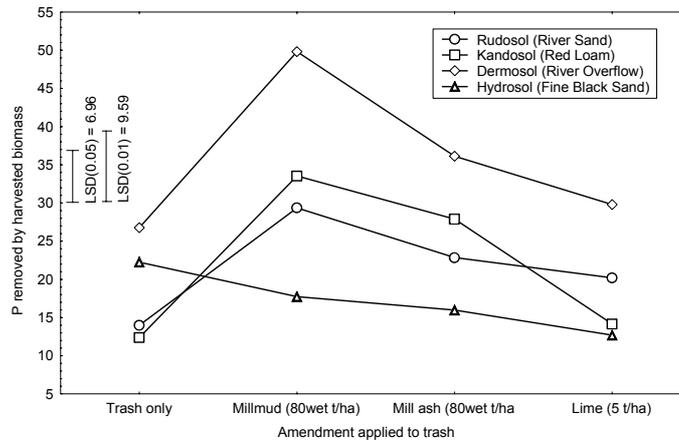


Figure 49 Interactive effect of soil type and amendments applied to the trash on P removed by the harvested biomass in Experiment 5

In Experiment 6, the highest mean dry matter yield was attained on the ferrosol (Figure 50). This was expected, as it is naturally a high yield potential soil. The yields obtained on the other two local Bundaberg soils (kandosol (Oakwood) and dermosol (Kepnock)), were in line with the general yield potential ranking. Dry matter yield on the rudosol was almost identical to that obtained on this soil under tropical conditions (Experiment 5).

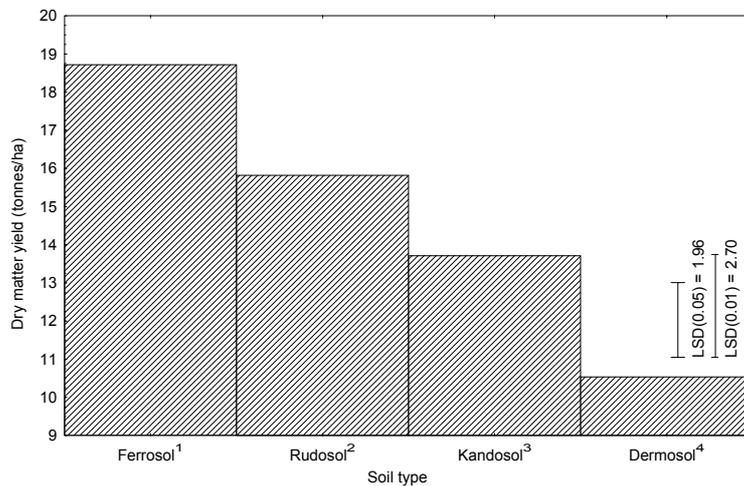


Figure 50 Mean dry matter yield values associated with each of the soil types included in Experiment 6

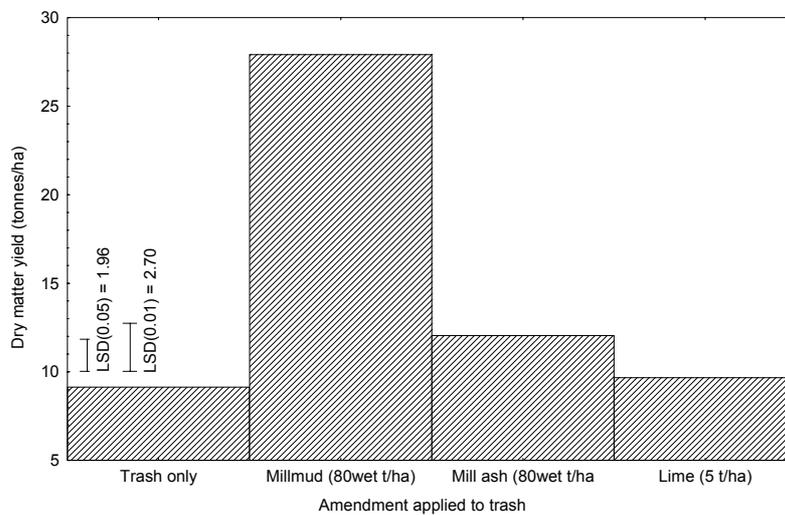


Figure 51 Mean dry matter yield values associated with the amendments applied to the trash in Experiment 6

The mean dry matter yield values associated with the amendments applied to the trash in Experiment 6 (Figure 51) indicated that, as in the case of the Macknade experiment, the largest response was due to the application of millmud (80 wet t/ha). Although the application of mill ash (80 wet t/ha) also resulted in a significant increase ($P < 0.05$) in dry matter production compared to the unamended trash treatment, lime (5 t/ha) applied to the trash blanket did not result in significantly different yields. This was surprising, as lime

applied to trash had increased decomposition of trash layer in all previous experiments and had resulted in increased yields in Experiment 5.

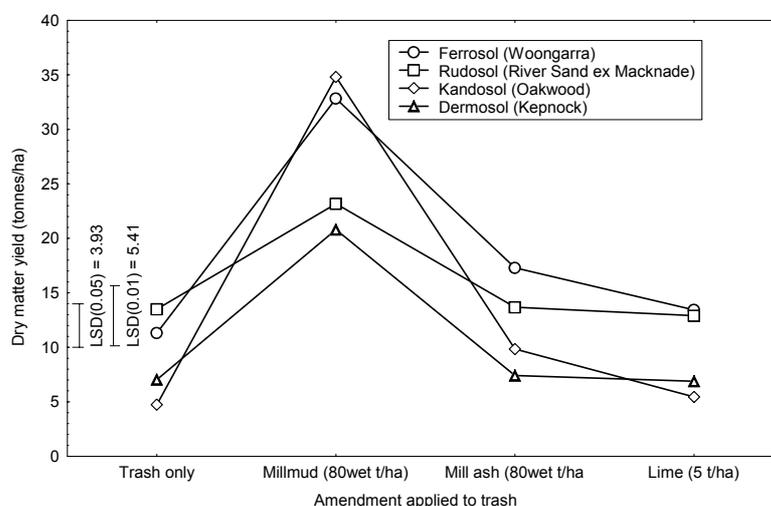


Figure 52 Interactive effect of soil type and amendment applied to the trash on dry matter yield in Experiment 6

Table 45 Dry matter yield data relating to environment (subtropical versus moist tropical) and amendment applied to trash on a rudosol (common soil in Experiments 5 and 6)

Environment	Dry matter yield (t/ha)	Amendment added to trash	Dry matter yield (t/ha)
Moist tropical	16.48	Trash only	12.83
Subtropical	15.81	Millmud (80 wet t/ha)	20.80
		Mill ash (80 wet t/ha)	15.35
		Lime (5 t/ha)	15.63
Significance	NS	Significance	*
		LSD (< 0.05)	3.98
Interaction			
Significance	NS		

Unlike the Macknade experiment (Experiment 5), soil type and amendment applied to the trash resulted in a significant interactive effect ($P < 0.01$) on dry matter yield (Figure 52). In particular, millmud applied to the trash layer on the rudosol (River Sand) and the dermosol (Kepnock) resulted in significantly lower dry matter yields than those achieved on the ferrosol (Woongarra) and kandosol (Oakwood). However, the yields (rudosol and dermosol) were not substantially different to the mean dry matter yield value (22.5 t/ha) associated with the millmud application to trash in Experiment 5. Comparisons of yields obtained in the two experiments was possible due to the inclusion of the rudosol (River

Sand ex Macknade) in both trials (Table 45). Yields associated with the River Sand were unaffected by the environment (moist tropical climate at Macknade versus subtropical conditions at Bundaberg), but affected markedly by millmud applications. No interactive effects were apparent.

Nitrogen removal by the harvested biomass, in terms of soil type and amendment applied to the trash (Figure 53), and the interaction between these two factors (Figure 54) reflected the trends that occurred in dry matter yields (Figures 50-52). These general trends also occurred in relation to K removal by the harvested biomass (Figures 57 and 58). Although P removal in relation to the amendments applied (Figure 55) also reflected the general dry matter yields (Figure 51), the P removal in the millmud treatment differed according to soil type. That P removed by the harvested biomass from the ferrosol was lower than that removed from the rudosol (Figure 56) is probably due to differences in P-sorbing characteristics.

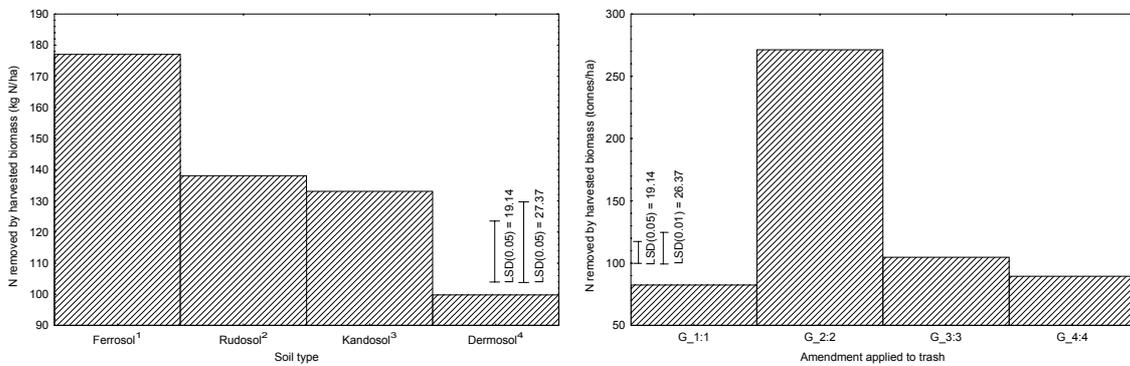


Figure 53 N removed by the harvested biomass plotted against soil type and amendments applied to the trash in Experiment 6

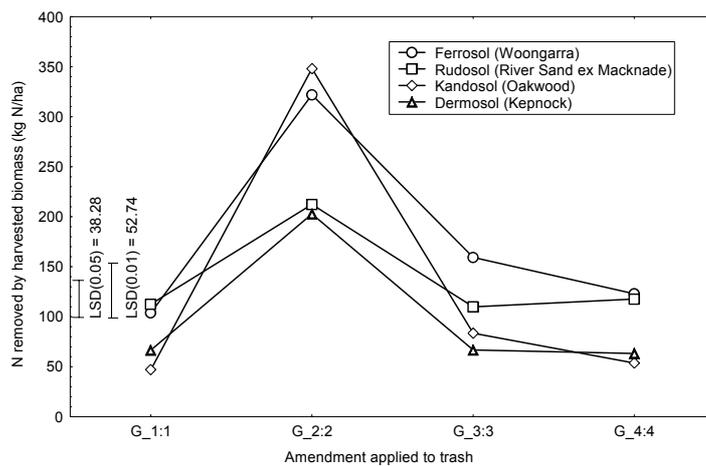


Figure 54 Interactive effect of soil type and amendments applied to the trash on N removed by the harvested biomass in Experiment 6

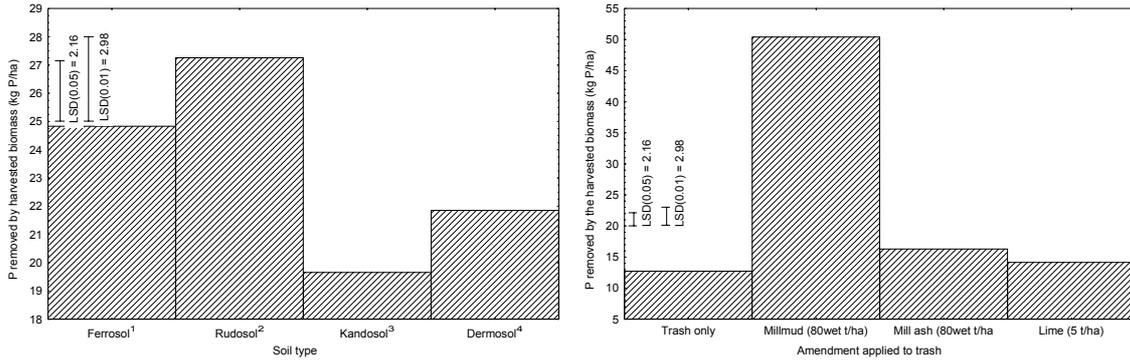


Figure 55 P removed by the harvested biomass plotted against soil type and amendments applied to the trash in Experiment 6

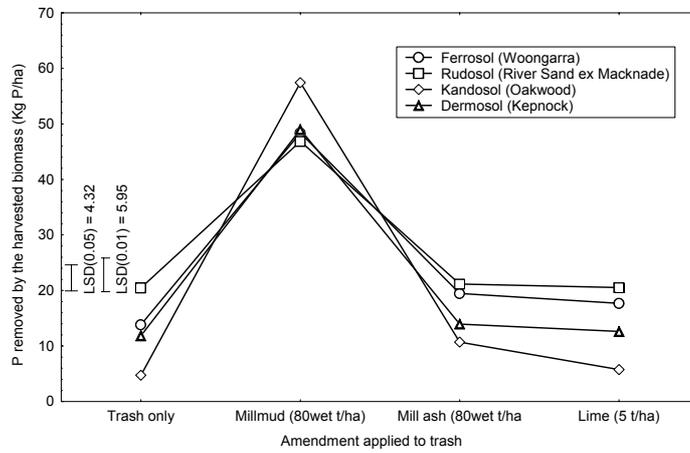


Figure 56 Interactive effect of soil type and amendments applied to the trash on P removed by the harvested biomass in Experiment 6

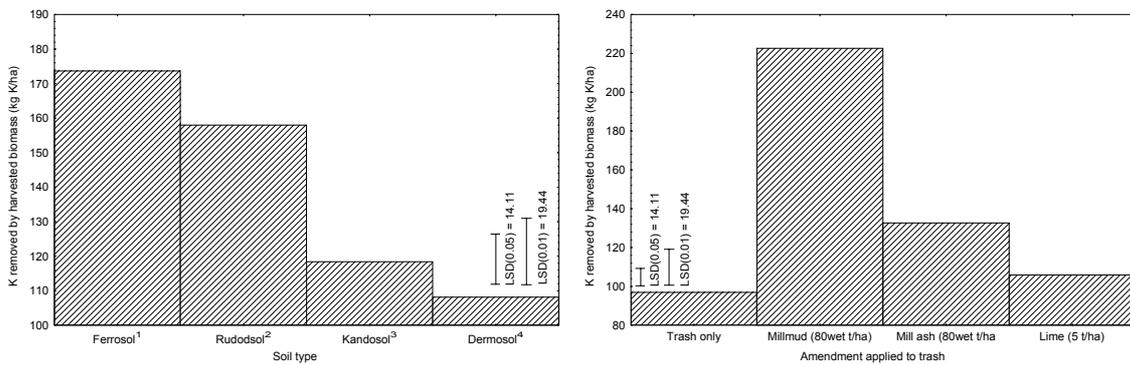


Figure 57 K removed by the harvested biomass plotted against soil type and amendments applied to the trash in Experiment 6

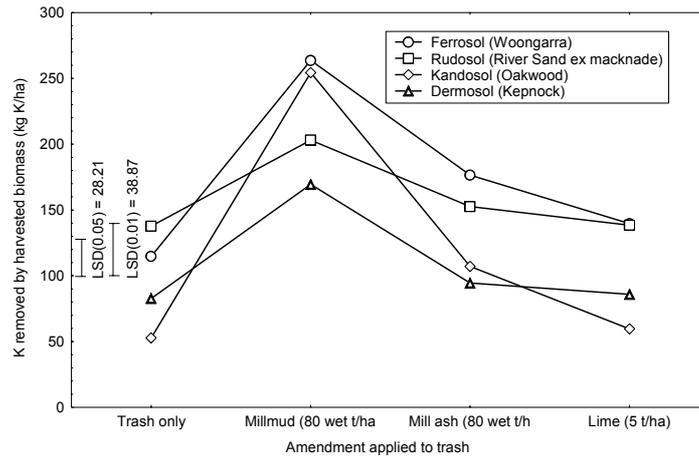


Figure 58 Interactive effect of soil type and amendments applied to the trash on K removed by the harvested biomass in Experiment 6

The differences noted between the two trials are best explained by the differences in the amount of water that moved through the pots. As both experiments were conducted on external seedling benches, the Macknade experiment would have benefited from both the irrigation and the relatively high rainfall in the moist tropical conditions (596 mm of rainfall was recorded at Macknade over the duration of the experiment). At Bundaberg, only 203 mm of rainfall was recorded during the trial period and, hence, much of the water was from the automatic irrigation system.

7.4 Conclusions

Calcium-based amendments such as lime and mill mud have the potential to accelerate the decomposition of sugarcane trash. Importantly this decomposition appears to be most effective when the amendments, particularly lime, millmud and mill ash, are applied to the surface of the trash layer. Lime applied beneath the trash and incorporated in the soil did not enhance the decomposition process. When comparing the effectiveness of the amendments in assisting decomposition, millmud repeatedly outperformed the other amendments. Whereas other amendments such as $\text{Ca}(\text{OH})_2$ and KOH did have the ability to enhance the decomposition process to some extent, they are not considered suitable for widespread use in the industry due to their corrosive nature. Amendments such as gypsum and MgO did not contribute to increased trash decomposition.

Importantly, the enhanced trash decomposition that occurred due to lime millmud, mill ash and lime applied to trash in pots under semi-controlled conditions was also found to be applicable in the field. The differences in the rates of decomposition in the simulated subtropical and moist tropical conditions suggest that environmental conditions involving temperature and the amount of water (rain and/or irrigation) are important factors within this trash/ameliorant system.

Lime applied to the surface of the trash layer did have the ability to ameliorate acid soil conditions. In some soils lime applied to the trash blanket resulted in higher pH values in

the subsurface soil layers than when lime was applied in the absence of trash. This apparent synergistic effect has important implications for certain soils, especially when minimum tillage is practiced. Importantly, millmud applications to the surface of the trash layer also resulted in some amelioration of the subsurface layers.

Although millmud and mill ash applications to trash resulted in significantly higher amounts of N, P and K removed by the harvested biomass, these elevated values could either have been the result of the nutrients within the amendments themselves, or due to enhanced decomposition of the trash, or a combination of both of these effects. The effect on nutrient uptake by sugarcane when lime was applied to trash was less apparent. Environmental effects (water, temperature and soil properties) are considered important factors in relation to the overall nutrient availability in these amended trash systems

8.0 TRASH DECOMPOSITION AND C AND N RELATIONSHIP

8.1 Introduction

In GCTB sugarcane systems, as in any cropping system involving retention of harvest residues, it is important to know not only the inputs of organic matter and nutrients in the residues, but how these residues interact with the soil-plant environment. Although this has been dealt with to some extent in Section 7, a more targeted approach within a 'micro' investigation system under more controlled conditions was deemed useful. Knowledge of the decomposition characteristics of the residues is fundamental to understanding their effects in the field.

It is also pertinent to know what factors affect trash decomposition, and how trash decomposition affects other soil properties and processes. Apart from the results of experiments reported earlier in this report, these subjects that have received little research attention, and much remains to be understood.

This section describes two experiments undertaken to address some of these uncertainties and determine whether small-scale pot experiments (using pots with < 20% of the surface area of those used in the experiments reported in Section 7) can be used to investigate some of these. The first experiment was done to gain basic information on decomposition of sugarcane trash in controlled conditions, and to help choose experimental conditions for future decomposition studies. The second experiment was done to measure the effects of mill mud, mill ash, urea, and sugarcane trash on soil C and N mineralisation using this type of experimental methodology.

8.2 Measuring trash decomposition in controlled conditions

Measurements of crop residue decomposition made under field conditions can sometimes be difficult to interpret due to variations in climate, soil, crop, and management factors. As such, experimentation under standardised (laboratory or glasshouse) conditions is often a practicable alternative way to measure decomposition and determine controlling factors.

Ideally, the conditions under which crop residues decompose in the field should be simulated in controlled environment studies. This may, however, be difficult to achieve, because experimentation on a small scale with limited resources requires the researcher to make compromises. It is usually necessary to dry the residue and to reduce the size of residue fragments (by chopping or grinding) to facilitate subsampling and handling. In many decomposition experiments, factors such as the size to which the residue is cut, whether the residue is fresh or dried, the placement of the residue in or on the soil, and the soil water content, vary from the field situation. The extent to which these factors affect decomposition of sugarcane trash is unknown, but studies with residues of other crops suggest that all these factors may be important (Parr and Papendick 1978; Christensen 1986; Bremer *et al.* 1991; Reddy *et al.* 1994; Andr n *et al.* 1995; S rensen *et al.* 1996; Dev vre and Horw th 2000).

This study was done to gain basic information on decomposition of sugarcane trash in controlled conditions using small-scale investigative units, and to help choose experimental conditions for future decomposition studies.

The objective of the work was to determine how decomposition of sugarcane trash is affected by: (1) whether the trash is green or senesced; (2) whether the trash is fresh or oven-dried; (3) whether the trash is chopped or ground; (4) whether the trash is on the soil surface or incorporated into the soil; and (5) whether the soil is wet or dry.

8.2.1 Methods

Experimental

Soil for the experiment was collected from the 0-10 cm depth of a sugarcane field at the Central Sugar Experiment Station, Mackay, Queensland (21.10 S, 149.07 E). The soil had contents of total C, total N, and inorganic N of 1.08%, 0.05%, and 7 $\mu\text{g/g}$, respectively. Particle size distribution was 56, 27, 17% (sand, silt, clay). The soil was dried, crushed and sieved (< 2 mm) before use. Trash was collected from mature sugarcane (cultivar Q124) growing in the field at the same location. Within 2 h of collection, the trash was separated into green leaves and senesced leaves, and was either oven-dried (65 C) or refrigerated (6 C) for 28 h. The trash was then either chopped with scissors (< 40 mm long) or passed through a cutter grinder (20-60 mm long x 3-6 mm wide). Soil and trash were sampled for analyses.

Sufficient plastic pots (100 mm diameter, sealed at the base) were prepared containing 250 g dry soil. Trash (equivalent to 4 g oven-dry) was either mixed with the soil or placed on the surface. Masses of soil and trash used in this experiment were about 4% of that used in the investigations reported in Section 7. Pots that contained unamended soil were also included. The soils were made up to the either 40% or 80% of saturation (i.e. 13% or 26% water content) by addition of deionised water. Table 46 shows the experimental treatments. The pots were placed inside heavy-duty zip-seal plastic bags (left open). Wire frames were placed in the pots, so that vials could be inserted for collection of CO₂. The pots were placed in the dark at a constant temperature of 25 C. Water was added weekly to maintain the original soil water contents.

Table 46 Experimental treatments

Factor	Level 1	Level 2
Trash fraction	Green	Senesced
Trash water content	Fresh	Dried
Trash fragment size	Chopped	Ground
Trash placement	Surface	Incorporated
Soil water content	Wet (80-90% saturation)	Dry (40-50% saturation)

Carbon dioxide was collected from the pots on 16 occasions over the next 8 months, using alkali traps (vials containing 20 mL of 1M sodium hydroxide were placed in the frames, and the plastic bags sealed for 2-3 d). At the end of the experiment, the contents of the pot were divided vertically, one half for retrieval of remaining trash, the other half for chemical analyses.

The experiment was designed as a ($\frac{1}{2} \times 2^5$) fractional factorial (Box and Hunter 1961) with five replicates, i.e. not all combinations of factors and levels were present. This design enabled the main effects and 2-way interactions to be measured, while limiting the samples to a manageable number.

Analytical methods

Soil inorganic N (ammonium + nitrate) was determined by extracting fresh soil in 2M potassium chloride, followed by automated colorimetric analysis of the extracts (Rayment and Higginson 1992, method 7C2). Total N and C were determined in dried soil (< 2 mm) and trash (< 1 mm) using a Leco combustion analyser. Carbon dioxide in the sodium hydroxide traps was measured by titration against dilute hydrochloric acid (Zibilske 1994). Soil pH and electrical conductivity were measured in 1:5 soil-water extracts (Rayment and Higginson 1992, method 3A1). Trash soluble C was extracted by shaking 1 g dry trash with 40 mL of deionised water (at room temperature) for 1 h, then filtering the extract. The total C concentration of the extract was determined by ICP. Residual trash was retrieved from the soil by washing the sample in approximately 3 L water containing 2 mL of Decon detergent then rinsing on a 2 mm sieve under running water. The trash was then dried at 65°C.

Calculations and statistical analysis

Cumulative CO₂-C evolved during the experiment (ΔC) was estimated from the area under the curve of CO₂ accumulation against time. Mineralisation of C due to trash (ΔC_T) was calculated as ΔC from trash-amended soil, less ΔC from unamended soil. Loss of trash-C was calculated as ΔC_T expressed as a percentage of trash-C applied.

Apparent net mineralisation or immobilisation of N (ΔN) was calculated as soil inorganic N at the end, less soil inorganic N at the start of the experiment. Apparent net mineralisation or immobilisation of N due to trash (ΔN_T) was calculated as ΔN from trash-amended soil, less ΔN from unamended soil. Apparent net mineralisation or immobilisation of N per unit of C mineralised was calculated as $\Delta N / \Delta C$ for unamended soil (ΔNC), and $\Delta N_T / \Delta C_T$ for trash-amended soil (ΔNC_T).

The significance of treatment effects (main effects and 2-way interactions) was measured by Analysis of Variance (ANOVA) for the appropriate fractional factorial design. Differences between some individual treatment combinations were also assessed by simple one-way ANOVA. Least significant differences (LSD) were calculated when ANOVA indicated significant effects ($P < 0.05$).

8.2.2 Results

Trash quality

Total C did not differ significantly among trash fractions or trash treatments (mean 36%). Green leaves contained 1.3% N and had a C:N ratio of 29; senesced leaves contained 0.4% N and had a C:N ratio of 93. This was not affected by trash drying or cutting treatments (Table 47). Soluble C in the trash was 3-10%, and was greater in green than in senesced leaves. The grinding treatment increased soluble C in both green and senesced leaves (Table 47).

Trash dry matter and C loss

Evolution of CO₂ from unamended soil was small compared with that from trash-amended soils. Cumulative CO₂ (ΔC) at the end of the experiment averaged 175 mg C/pot in the unamended soils, and 700-1200 mg C/pot in the trash-amended soils. In unamended soils, there was a consistent, but usually non-significant, trend for ΔC to be slightly greater in wet soil than in dry soil (Figure 59).

Table 47 Trash C and N at the start of the experiment (presented as % of dry matter)

Trash	Total N (%)	Total C:N ratio	Soluble C (%)
Green, fresh, chopped	1.32	27	9.3
Green, fresh, ground	1.25	27	10.3
Green, dried, chopped	1.34	30	8.9
Green, dried, ground	1.26	31	10.0
Senesced, fresh, chopped	0.44	87	3.6
Senesced, fresh, ground	0.36	88	4.6
Senesced, dried, chopped	0.45	91	3.1
Senesced, dried, ground	0.39	91	4.0
LSD ($P < 0.05$)	0.11	6	0.8

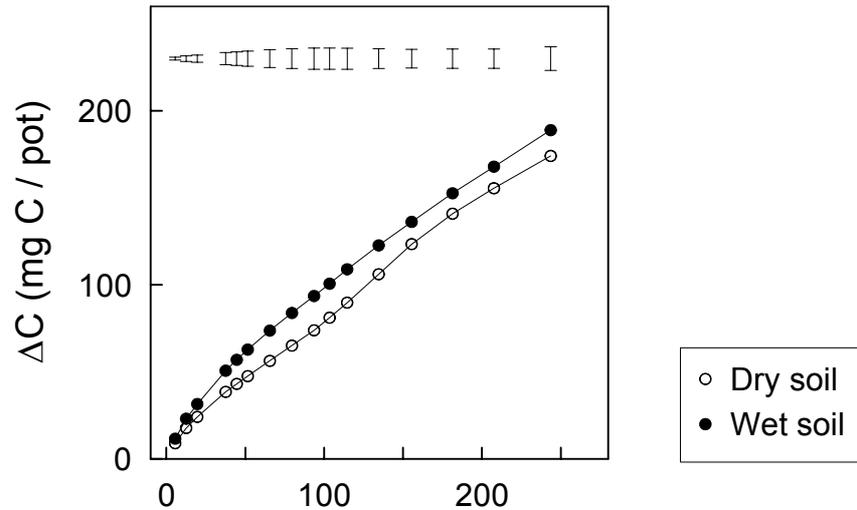


Figure 59 Cumulative CO₂ evolution from unamended soils (ΔC). Bars are LSD ($P = 0.05$) for unamended soil only

Trash decomposition measured as CO₂ evolution (ΔC_T), was initially faster in green leaves than in senesced leaves (until day 20), but this trend was reversed from day 94 onwards, and by the end of the experiment (day 244) ΔC_T was not significantly different between trash fractions (Figure 60).

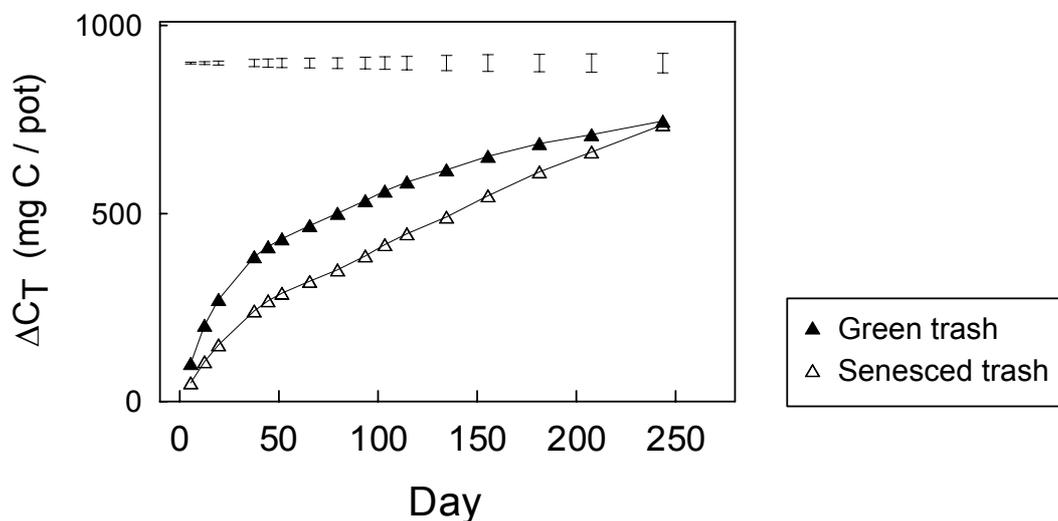


Figure 60 Cumulative CO₂ evolution from green and senesced trash (ΔC_T). Bars are LSD ($P = 0.05$) for main effect

Trash water content had no effect on ΔC_T (data not shown).

Trash placement and trash particle size also had some early transient effects on ΔC_T . During the first 20-38 d, senesced leaves decomposed more rapidly when incorporated

with the soil than when placed on the surface; and green leaves decomposed more rapidly on the surface than when incorporated (data not shown). Ground trash decomposed faster than chopped trash only when placed on the surface and only for the first 13 d (data not shown). Trash particle size did not affect final ΔC_T .

At most sampling times and at the end of the experiment, ΔC_T showed a highly significant interaction between trash placement and soil water. In wet soil, ΔC_T was greater with trash on the surface; in dry soil, ΔC_T was greater with trash incorporated (Figure 61).

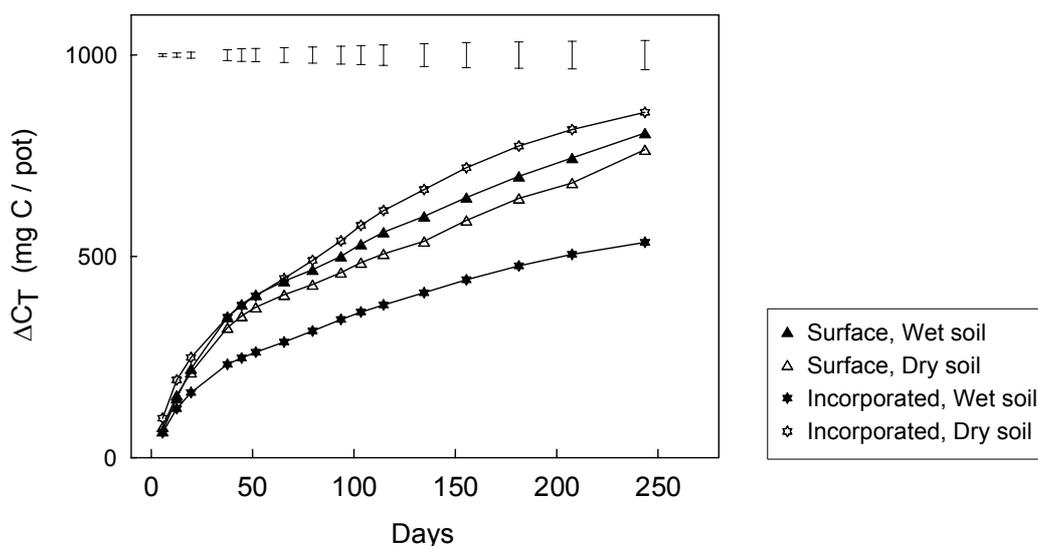


Figure 61 Interacting effects of trash placement and soil water content on cumulative CO₂ evolution from trash (ΔC_T). Bars are LSD ($P = 0.05$) for interaction

At the beginning of the experiment only, ΔC per day was positively correlated with both the original C:N ratio and soluble C content of the trash ($r = 0.8, 0.6, 0.6$ for days 6, 13, 20, respectively, $P < 0.01$, $n = 16$).

Trash decomposition measured as dry matter (DM) loss was more variable among replicates than ΔC , but generally showed similar trends (Figure 62). Trash fraction and trash water content had no effect on DM loss. There was a significant interaction between trash placement and soil water content. In wet soil DM loss was greater with surface-applied trash, and in dry soil DM loss was greater with incorporated trash. DM loss tended to be greater when trash was ground than when it was chopped, but this is not considered to be a true treatment effect, but rather a reflection of the relative ease with which trash could be retrieved from the chopped treatments compared to the ground treatments. Percentage decomposition of trash during the experiment was 32-89% (mean 60%) when measured as DM loss, and 35-66% (mean 50%) when measured as C loss from CO₂ data (Figure 62). Correlation between DM loss and C loss gave $r = 0.51$ ($P < 0.001$, $n = 80$).

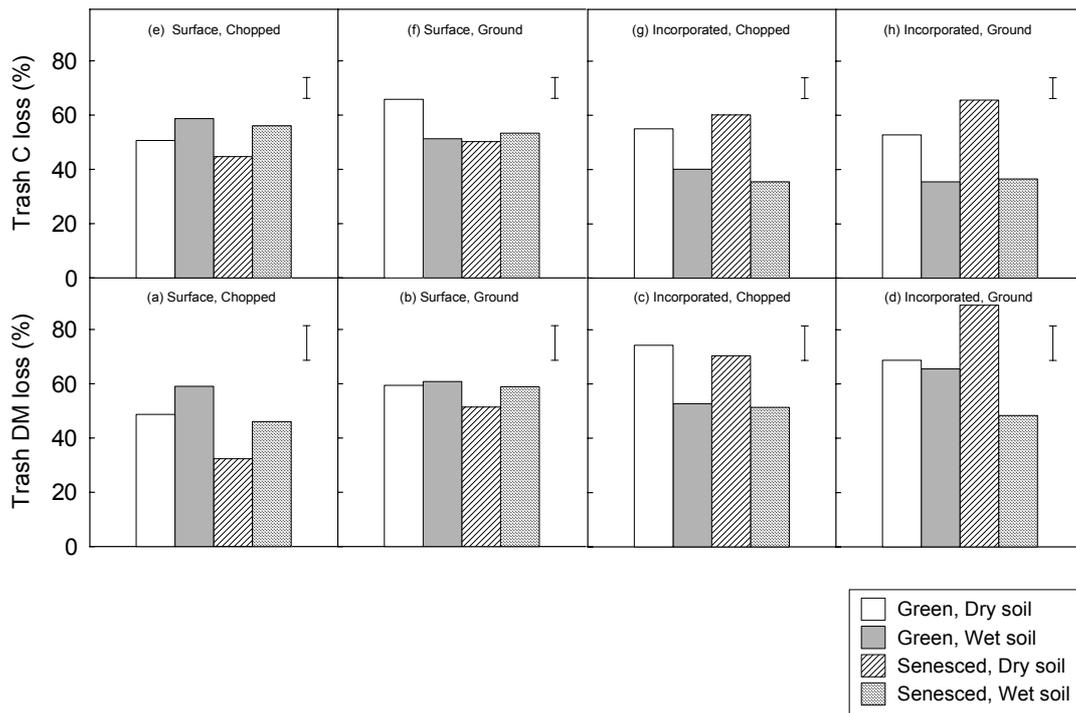


Figure 62 Dry matter (DM) and C loss from trash at the end of the experiment. Bars are LSD ($P = 0.05$) from one-way ANOVA

N mineralisation and immobilisation

Apparent net N mineralisation in unamended soil (ΔN) averaged 16 mg N/pot. Apparent net mineralisation (positive value) or immobilisation (negative value) of N due to trash (ΔN_T) ranged from -11 to 9 mg N/pot. It was significantly affected by trash fraction (green, mean 5.3 mg N/pot > senesced, mean -9.1 mg N/pot) (Figure 63a-c). Generally, ΔN_T was greater for surface than for incorporated trash (except for chopped trash in dry soil). Neither trash water content (data not shown) nor trash particle size, had a consistent effect on ΔN_T . There was a trend for ΔN_T to be lower in wet soil than in dry soil (Figure 63 a-c).

Apparent net N mineralisation expressed on the basis of C mineralisation (ΔNC) was 12.6 mg N/g C for unamended soil. The corresponding values for trash (ΔNC_T) were 8 and -17 mg N/g C for green trash and senesced trash, respectively. Again, ΔNC_T was generally greater for surface than for incorporated trash, except for chopped trash in dry soil. Trash water content (data not shown) and trash particle size, had no consistent effect on ΔNC_T (Figure 63 d-f). With senesced trash that was incorporated, or ground and surface-applied, ΔNC_T was lower in wet soil than in dry soil. With green trash, ΔNC_T tended to be lower in wet soil than in dry soil when the trash was incorporated, but soil-water content had no effect when the trash was surface-applied (Figure 63 d-f).

Correlation analysis showed no relationship between ΔN_T and ΔNC_T (with data separated by trash fraction, or with pooled data).

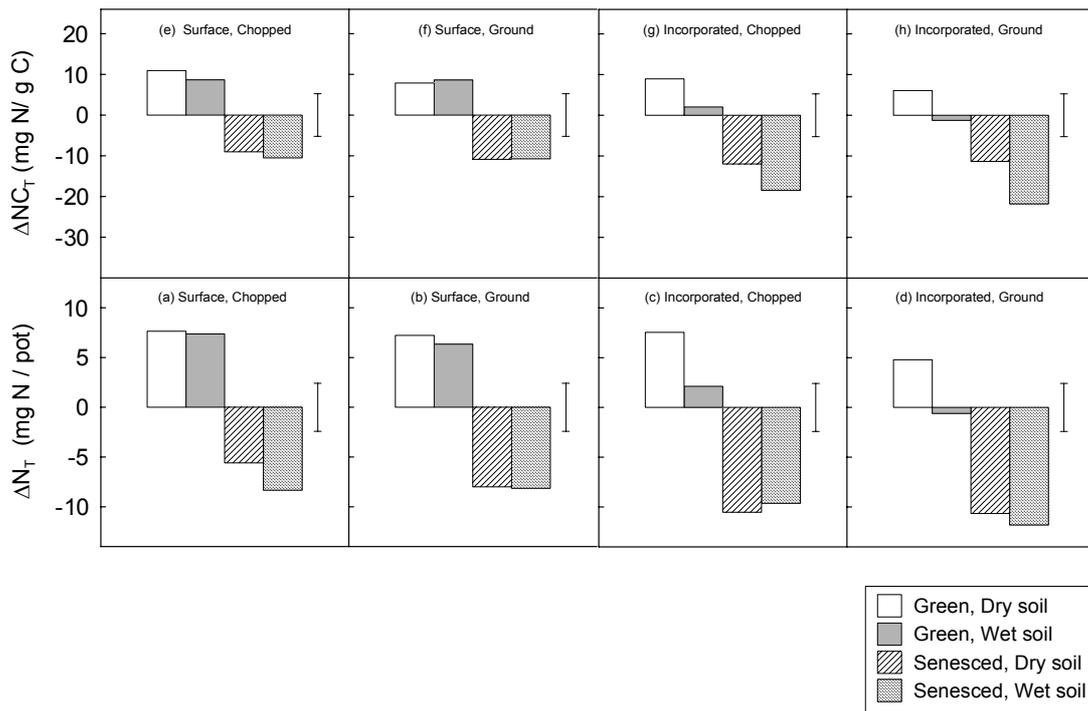


Figure 63 Apparent net N mineralisation from trash (ΔN_T), and apparent net N mineralisation per unit of C mineralisation from trash (ΔNC_T), at the end of the experiment. Bars are LSD ($P = 0.05$) from one-way ANOVA

8.2.3 Discussion

The aim of this work was to measure the effects of the experimental treatments on decomposition of trash. The conclusions to be drawn from the results depend on whether ‘decomposition’ is being measured as DM loss, C mineralisation, or apparent net N mineralisation / immobilisation.

Measuring decomposition as DM loss may appear simple and inexpensive in comparison with C and N measurements, but it was somewhat unreliable in this small-scale experiment. Variability among replicates was large, because of the difficulty in retrieving trash from the soil at the end of the experiment, particularly when the trash had been ground. Division of the pot contents before trash retrieval would have contributed to the variability. Decomposition was probably overestimated because particles smaller than 2 mm were not retrieved. It could be expected that soils of higher clay content than that used in this experiment would exacerbate these problems. Notwithstanding these deficiencies, the DM measurements generally agreed with the CO_2 measurements (except with ground trash). DM loss suggested that trash fraction and trash water content had no effect on decomposition. The effect of trash placement depended on soil water content; in wet soil DM loss was greater with surface-applied trash, and in dry soil DM loss was faster with incorporated trash. To improve the usefulness of DM loss as an indicator of decomposition in similar controlled environment studies, the trash should not be ground or shredded, the trash fragment size should be > 4 cm, and the pot contents should not be divided.

Measurements of decomposition as C mineralisation (CO_2) were less variable within treatments than the DM loss measurements. Thus, they can be considered more reliable, and had the advantage of allowing decomposition to be measured non-destructively over time. Decomposition may have been underestimated, since the alkali absorption method used in this experiment has been shown to slightly underestimate CO_2 evolution from the soil surface (Freijer and Bouten 1991). The relative differences among treatments, however, would not have been affected. The weakness of the correlation between CO_2 -C loss and DM loss can be attributed mainly to the experimental error associated with the latter.

Decomposition as measured by ΔC_T was temporarily influenced (for up to 5 weeks) by whether the trash was green or senesced, and whether the trash was chopped or ground, but neither trash fraction nor trash particle size affected the final decomposition at day 244. The early increase in ΔC_T from green trash could be attributed to its greater soluble C content and smaller lignin content (Thorburn, unpubl. data). The temporary increase in ΔC_T from ground trash was probably due to liberation of soluble C by grinding, and improved soil contact because of the smaller particle size. Similar trends have been reported by Bremer *et al.* (1991) for decomposition of straw and green material from lentil. Decomposition as measured by C mineralisation was not affected by whether the trash was fresh or oven-dried. Decomposition was affected by both trash placement and soil water content and, more importantly, there was a strong interaction between these factors; in wet soil trash decomposed more rapidly on the surface (because of improved aeration), and in dry soil trash decomposed more rapidly when incorporated (because of improved water availability and soil contact). So, in decomposition experiments involving CO_2 measurements, soil water content and trash placement need to be carefully controlled. In short-term experiments of several weeks, the proportion of green and senesced material, and its particle size needs to be controlled, but in long-term experiments, variations in trash fraction and particle size would be less important. Oven-drying the trash is unlikely to affect CO_2 measurements, and allows greater flexibility in sample handling.

Decomposition as measured by apparent net N mineralisation or immobilisation was most strongly influenced by trash fraction, with net mineralisation predominating in the green trash and net immobilisation predominating in the senesced trash, in accordance with the initial C:N ratio of the trash. Apparent N mineralisation or immobilisation was not affected by whether the trash was fresh or oven-dried, or whether it was chopped or ground.

The ratio of net N mineralised or immobilised per unit of C mineralised (ΔNC) from a residue gives an indication of the balance between mineralisation and immobilisation of N, assuming no loss of N, and reflects the C and N composition of the original residue. Thus, ΔNC_T would be expected to vary little within green trash and within senesced trash treatments. This was the case in dry soil, indicating that trash placement did not affect the balance between N mineralisation and immobilisation. In wet soil, however, ΔNC_T was lower when trash was incorporated than when it was surface-applied, indicating either (a) increased immobilisation with incorporated trash, or (b) N loss with incorporated trash. Option (a) is unlikely as the decomposition substrate did not change. Option (b), on the other hand, is highly probable: incorporation of labile organic matter into wet soil can lower oxygen tension and stimulate denitrification (Alexander 1977).

Thus, in decomposition experiments where apparent net mineralisation / immobilisation of N is measured, it would be very important to control the proportion of green and senesced material, as well as soil water content and trash placement. In particular, N mineralisation may be underestimated (or immobilisation overestimated) when trash is incorporated into wet soil. Oven-drying, grinding, or chopping the trash are not likely to affect N mineralisation / immobilisation.

This study illustrates the usefulness of combining measurements of C and N mineralisation. The lack of correlation between the two show that they measure different aspects of decomposition, and the C measurements are a valuable aid to interpretation of the N measurements. The interacting effects of trash placement and soil water content shown by DM loss and C and N mineralisation have practical implications for trash management in the field. If the soil is predominantly fairly dry, incorporation of the trash is likely to slightly increase DM loss without affecting net N mineralisation/immobilisation; if the soil is wet for extended periods, incorporation of the trash is likely to slow DM loss, and increase N loss from the field through denitrification.

8.3 Trash decomposition and effects of trash, mill mud, mill ash and urea on soil C and N

As indicated in Section 7, sugarmill by-products such as mill mud and mill ash have long been recognised as potentially valuable sources of soil nutrients. General application guidelines have been developed (Calcino 1994) based on nutritional considerations, but in practice mill mud and ash are applied to canefields at various rates, according to factors such as cost, distance from the sugar mill, and convenience.

Field trials suggest that a typical application of 150 t/ha of mill mud or mill mud-ash mixture can provide around 100 kg /ha of plant-available N (around 20% of the total N content) in the year of application (Calcino 1994). How nutrient availability may be affected by the presence of cane trash, or other fertilisers such as urea, is not known. Nutrient release from mill mud or ash has not been measured directly.

The objective of this work was to measure, under standardised small-scale conditions (similar to those described in Section 8.2), the individual and interacting effects of mill mud, mill ash, urea, and sugarcane trash on soil C and N mineralisation.

8.3.1 Methods

Experimental

Soil was collected from the 0-15 cm depth of sugarcane fields at three locations: Herbert, Burdekin, and Mackay (Table 48). The Burdekin and Mackay soils were also used in the experiment described in section 4.3, and the Mackay soil was also used the experiment described in section 4.2. The soil was dried in an oven at 45°C for 6 d, then crushed to pass through a 2 mm screen, and sampled for analyses.

Sugarcane (cultivar Q124) was grown in 40-cm-diameter pots in a glasshouse for 1 year, fertilised with weekly applications of a mixed fertiliser including ¹⁵N-labelled ammonium sulfate. At maturity, the green and senesced leaves, were collected. They were dried in

an oven at 65°C for 2 d, shredded through a cutter-grinder (2-10 cm long and 0.2-1.0 cm wide), mixed thoroughly, and subsampled. The fragment size of the subsample was then reduced to < 4 cm by cutting with scissors.

Table 48 Soils used in the experiment (0-15 cm depth)

Parameter	Herbert	Burdekin	Mackay
Location	Unknown	147.23 E, 19.50 S	149.07 E, 21.10 S
Texture	Organic sand	Sandy clay loam	sandy loam
Particle size (%sand, %silt, %clay)	Not determined	53, 25, 22	56, 28, 16
Total N (%)	0.14	0.04	0.05
Total organic C (%)	2.76	0.53	0.95
C:N ratio	20	13	19
Inorganic N (µg/g)	26	11	8
Saturation water content (%)	26	42	40

Fresh mill mud and mill (bagasse) ash were obtained from Racecourse sugar mill, Mackay. It was dried in an oven at 45°C for 6 d. Clumps of dry mill mud were crushed to < 10 mm for use in the experiment. The ash was free-flowing with particles < 5 mm. Samples of ash and mud (500 g) were ground to pass through a 2 mm screen before being subsampled for analyses (Table 49).

Table 49 C and N in amendments used in the experiment. Results are % of dry matter

Measurement	Mill ash	Mill mud	Trash
Total C (%)	8.8	26.7	39.9
Total N (%)	0.06	1.22	0.68
Total C:N ratio	142	22	60
Soluble C (%)	0.38	2.87	1.94
Soluble N (%)	0.000	0.003	0.012
¹⁵ N atom %	-	-	7.04

Experimental units were prepared by mixing 300 g dry soil with the appropriate amendments and placing the mixtures in plastic pots (9-cm diameter) lined with plastic bags. Water was added to bring the mixtures to 70% of saturation. The equivalent application rates of the amendments were: trash 10 t DM/ha; mill mud 40 t DM/ha (equivalent to 160 t/ha wet); mill ash 40 t DM/ha; urea 180 kg N /ha (applied as 5 mL of 18.1 mg N /mL solution). The scale of the pots and mass of soils trash and amendment were about 5% of those used in the investigations reported in Section 7 of this report. The amendments were applied alone to the Herbert and Burdekin soils, and alone and in combinations to the Mackay soil, giving 20 treatments (Table 50). Four replicates of the treatments were prepared. The pots were placed inside heavy-duty zip-seal plastic bags

(left open). Wire frames were placed in the pots so that vials could be inserted for collection of CO₂. The pots were placed in the dark at a constant temperature of 25°C. Water was added weekly to maintain the original soil water contents. The pots were arranged as randomised complete blocks, and positions within blocks were rerandomised on three occasions.

Table 50 **Experimental treatments**

Treatment	Soil	Amendments	Total applied (mg/pot)			
			Total C	Total N	Soluble C	Soluble N
1	Herbert	Unamended soil	-	-	-	-
2	Herbert	Ash	1760	12	76	0.00
3	Herbert	Mud	5336	244	573	0.68
4	Herbert	Trash	1995	34	97	0.59
5	Burdekin	Unamended soil	-	-	-	-
6	Burdekin	Ash	1760	12	76	0.00
7	Burdekin	Mud	5336	244	573	0.68
8	Burdekin	Trash	1995	34	97	0.59
9	Mackay	Unamended soil	-	-	-	-
10	Mackay	Ash	1760	12	76	0.00
11	Mackay	Mud	5336	244	573	0.68
12	Mackay	Trash	1995	34	97	0.59
13	Mackay	Urea	0	90	0	90
14	Mackay	Ash + Urea	1760	102	76	90
15	Mackay	Mud + Urea	5336	334	573	90.68
16	Mackay	Trash + Ash	3755	46	173	0.59
17	Mackay	Trash + Mud	7331	277	670	1.26
18	Mackay	Trash + Urea	1995	124	97	90.59
19	Mackay	Trash + Ash + Urea	3755	136	173	90.59
20	Mackay	Trash + Mud + Urea	7331	367	670	91.26

Carbon dioxide was collected from the pots on 10 occasions over the next 6 months, using alkali traps (vials containing 20 mL of 1M sodium hydroxide were placed in the frames, and the plastic bags sealed for 2-3 d).

At the end of the experiment, the contents of each pot were mixed and sampled for analyses.

Analytical methods

Soil inorganic N (ammonium + nitrate) was determined by extracting 15 g of fresh soil in 2M potassium chloride, followed by automated colorimetric analysis of the extracts ((Rayment and Higginson 1992, method 7C2). Total N and C were determined in dried soil (< 2 mm) and trash (< 1 mm) using a Leco combustion analyser. Carbon dioxide in the sodium hydroxide traps was measured by titration against dilute hydrochloric acid (Zibilske 1994). Soil pH and electrical conductivity were measured in 1:5 soil-water extracts (Rayment and Higginson 1992, method 3A1). Trash-soluble C was extracted by shaking 1 g dry trash with 40 mL of deionised water (25°C) for 1 h, then filtering the extract. The total C concentration of the extract was determined by ICP-OES.

Calculations and statistical analysis

Cumulative CO₂-C evolved during the experiment (ΔC) was estimated from the area under the curve of CO₂ accumulation against time. Mineralisation of C due to the amendment (ΔC_A) was calculated as ΔC from amended soil, less ΔC from unamended soil. Apparent net mineralisation or immobilisation of N (ΔN) was calculated as soil inorganic N at the end, less soil inorganic N at the start of the experiment. Apparent net mineralisation or immobilisation of N due to the amendment (ΔN_A) was calculated as ΔN from amended soil, less ΔN from unamended soil. Apparent net mineralisation or immobilisation of N per unit of C mineralised was calculated as $\Delta N / \Delta C$ for unamended soil (ΔNC), and $\Delta N_A / \Delta C_A$ for amended soil (ΔNC_A).

The soil and single-amendment effects were tested by analysis of variance (ANOVA) for a two-way factorial, using treatments 1-12. Single and mixed-amendment effects in the Mackay soil were also tested by one-way ANOVA, using treatments 9-20. Least significant differences (LSD) were calculated when ANOVA indicated significant effects ($P < 0.05$).

8.3.2 Results

C mineralisation

In unamended soil, mineralisation of C (ΔC) was similar in the Burdekin and Herbert soils, and greater than in the Mackay soil (Figure 64).

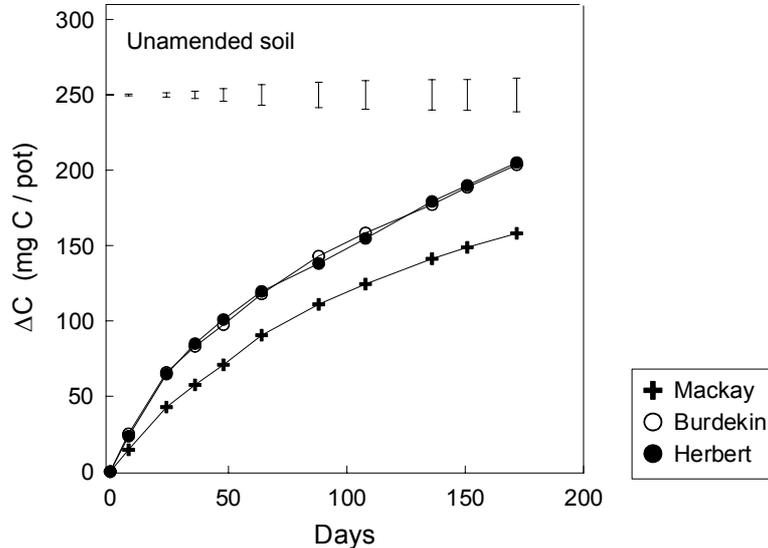


Figure 64 Cumulative CO₂ evolution (ΔC) in unamended soils. Bars are LSD ($P = 0.05$) for unamended soils only

Mill ash had no effect on C mineralisation in any of the soils (ΔC in ash-amended soils was not significantly different from unamended soils, Figure 65). Trash and mill mud produced large increases in ΔC , with mill mud mineralising around twice as much C as trash per pot (Figure 65). When C mineralisation was expressed per unit of amendment

dry matter applied (ΔC_A), however, C mineralisation decreased in the order trash > mill mud > mill ash (Figure 66). Most of the difference in ΔC_A between trash and mill mud was due to CO_2 evolved during the first 88 days (data not shown).

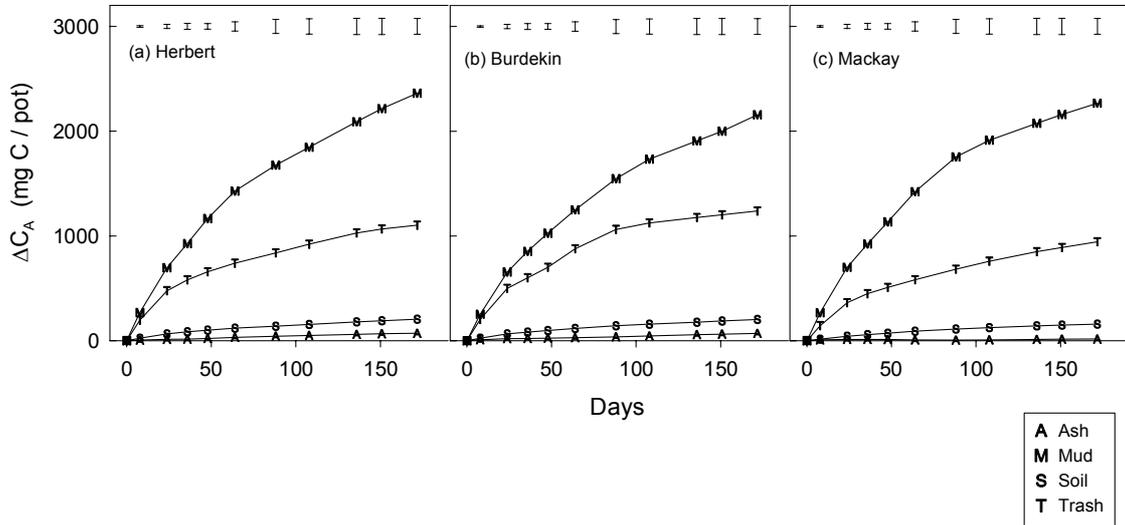


Figure 65 Cumulative CO_2 evolution from soils (ΔC) and amendments (ΔC_A). Bars are LSD ($P = 0.05$) for soil type x amendment interaction.

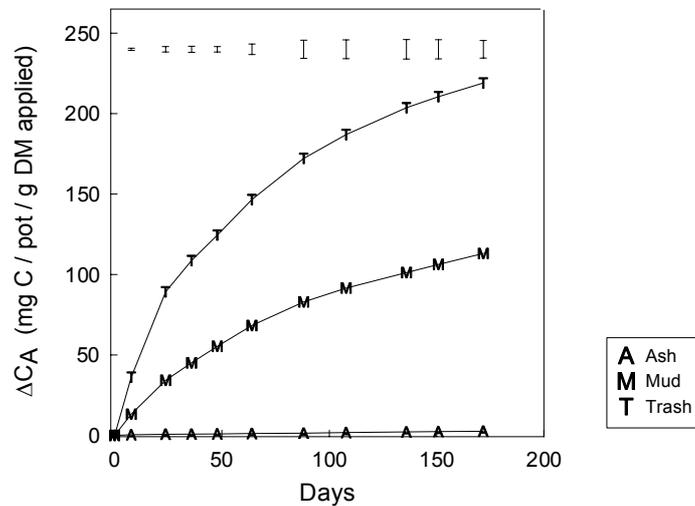


Figure 66 Cumulative CO_2 evolution from amendments (ΔC_A) per g dry matter. Bars are LSD ($P = 0.05$) for main effect

In the Mackay soil, urea applied alone had no effect on C mineralisation (Figure 67). Urea applied with trash caused a small transient increase in total ΔC_A (ΔC_A from trash + urea was significantly greater than ΔC_A from trash until day 24). Ash applied with trash slightly increased total ΔC_A (to day 108), the effect of which was evident till the end of the experiment. Urea and ash applied together with trash had a similar effect on ΔC_A as their combined effects when applied alone with trash (Figure 67). The greatest ΔC_A was

from trash + mud, and trash + mud + urea (Figure 67). The mixtures containing mill mud gave a slightly smaller ΔC_A at the end of the experiment than the total ΔC_A of the same amendments alone (Figure 70a). The other amendments gave similar ΔC_A values, whether mixed or alone. Throughout the experiment, ΔC (per day, or cumulative) was positively correlated with total C applied ($r \geq 0.82$, $P \leq 0.001$, $n = 80$, data not shown).

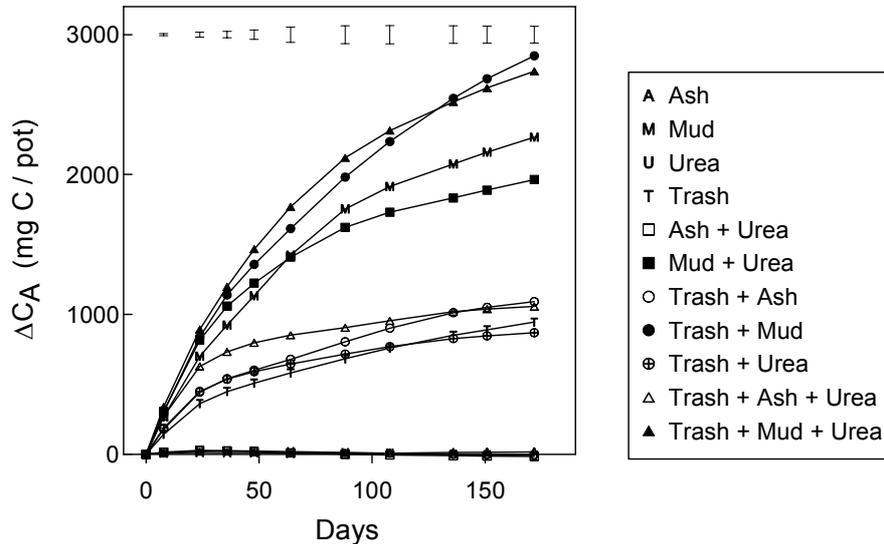


Figure 67 Cumulative CO₂ evolution (ΔC_A) in the Mackay soils. Bars are LSD ($P = 0.05$) from one-way ANOVA

N mineralisation

Apparent net mineralisation of N (ΔN) in unamended soil was greatest in the Herbert soil (14 mg/pot), and similar in the Burdekin and Mackay soils (6 and 7 mg/pot, respectively (Figure 68 a-c).

Mill ash had no effect on N mineralisation (ΔN) in the Herbert and Mackay soils, and a very small positive effect in the Burdekin soil (Figure 68 a-c). Trash alone immobilised approximately the same amount of N as was mineralised by the unamended soils. Mill mud alone mineralised similar amounts of N in all soils (mean 8 mg/pot) (Figure 68 a-c). The net immobilising effect of trash, net mineralising effect of mill mud, and net neutral effect of mill ash were more evident when ΔN_A was expressed per unit of amendment dry matter applied (Figure 68 d-f).

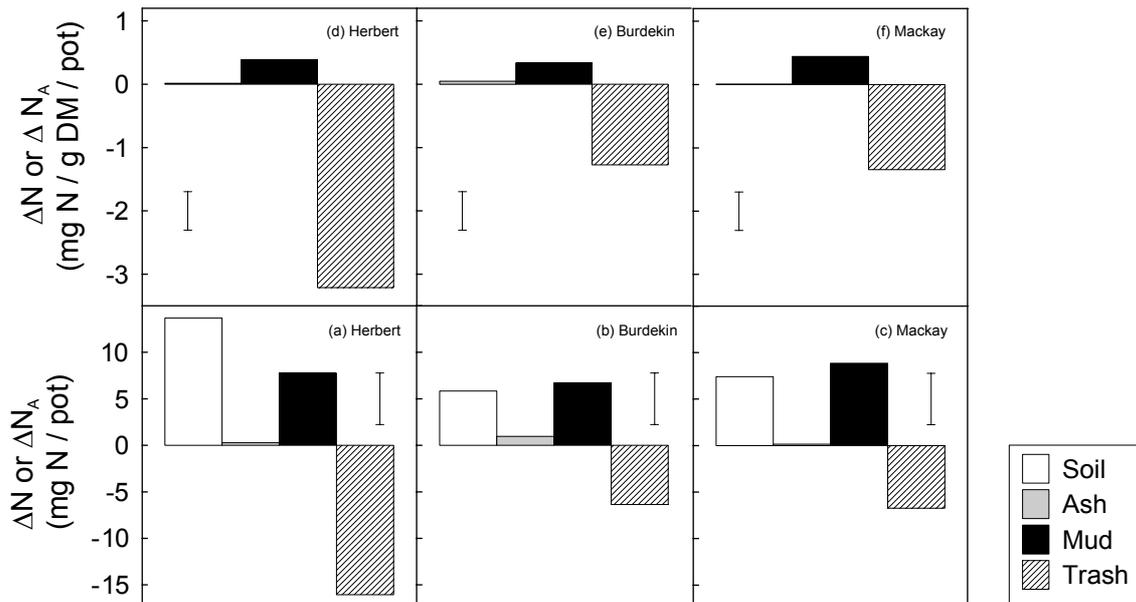


Figure 68 Apparent net N mineralisation / immobilisation from soils (ΔN) and amendments (ΔN_A), per pot and per unit dry matter (DM) applied. Bars are LSD (P = 0.05) for soil type x amendment interaction

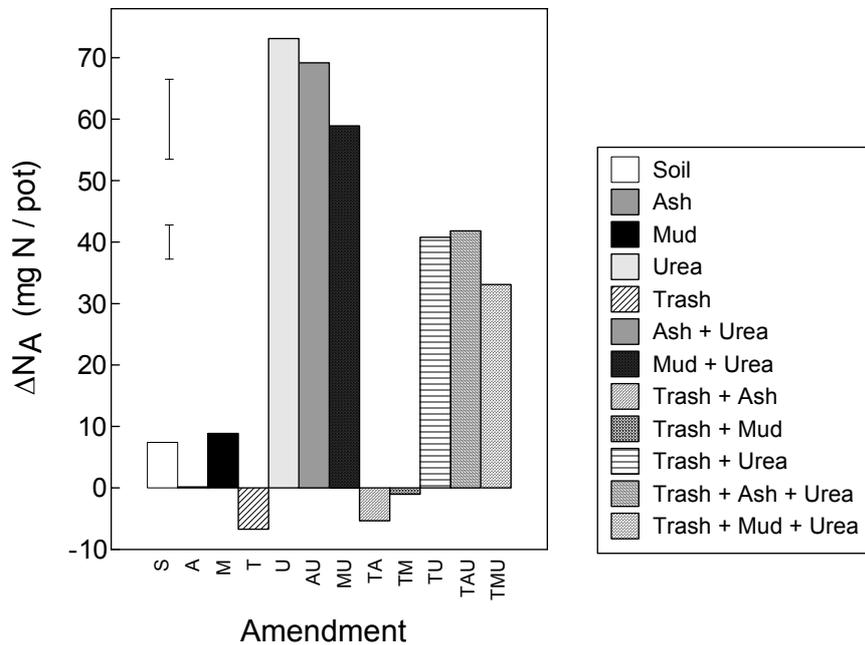


Figure 69 Apparent net N mineralisation / immobilisation from soil (ΔN) and amendments (ΔN_A) in the Mackay soil. Bars are LSD (P = 0.05) for soil type x amendment interaction (short bar) and one-way ANOVA (long bar)

In the Mackay soil, urea applied alone produced the greatest ΔN_A (74 mg N/pot). The ΔN_A was not affected when ash was applied with urea, but was reduced when mill mud and/or trash were applied with urea (Figure 69). Ash applied with trash resulted in ΔN_A similar to that from trash applied alone (Figure 69). Mill mud applied with trash tended to increase ΔN_A compared with trash alone (but the effect was not significant, one-way ANOVA).

The ΔN_A for amendment mixtures was less than the total ΔN_A for the same amendments alone, particularly for mixtures containing mill mud (Figure 70). The ΔN_A was not correlated with total N applied, the overall C:N ratio of the amendments, or with ΔC_A .

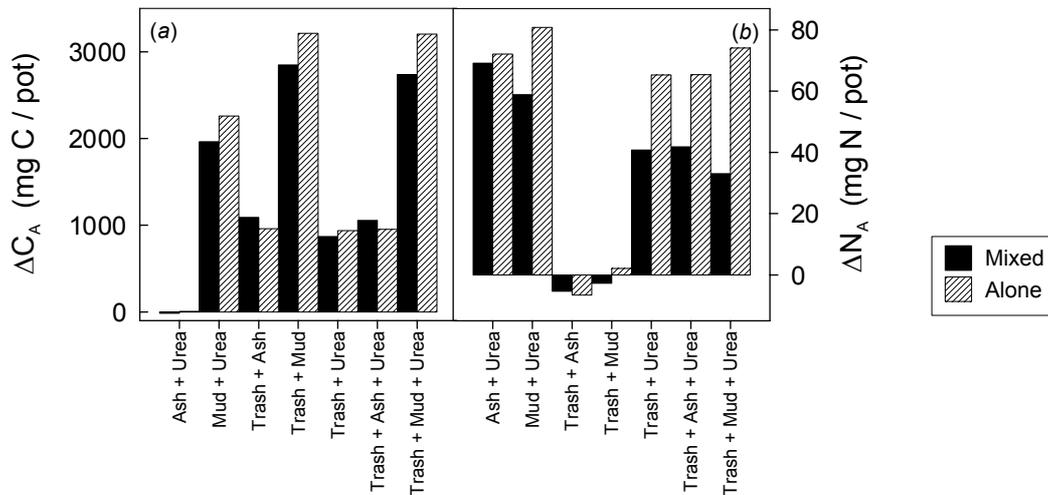


Figure 70 Cumulative CO_2 evolution (ΔC_A) and apparent net N mineralisation /immobilisation (ΔN_A) from amendments applied alone or in mixtures

Apparent net N mineralisation or immobilisation expressed on the basis of C mineralisation (ΔNC) was relatively high for unamended soils, with Herbert > Mackay > Burdekin (Figure 71). The corresponding measure for mill mud alone (ΔNC_A) was positive but very much smaller, and similar in the three soils. The ΔNC_A for trash alone was negative, and lower in the Herbert than in the Burdekin and Mackay soils (Figure 71). The trash + mud mixture gave a ΔNC_A close to zero. The trash + mud + urea mixture gave a considerably smaller ΔNC_A than the trash + urea, and the trash + urea + ash mixtures (Figure 71).

Total C and N

In all soils, total soil N was significantly increased by mill mud, but not by ash or trash (Figure 72 a-c). Urea did not significantly affect total N in the Mackay soil (data not shown). Total soil C was increased by mill mud in all soils, and by ash in the Burdekin and Mackay soils (Figure 72 d-f). For both total N and C, the mixtures of amendments in the Mackay soil conformed to the trends shown by the individual amendments (data not shown).

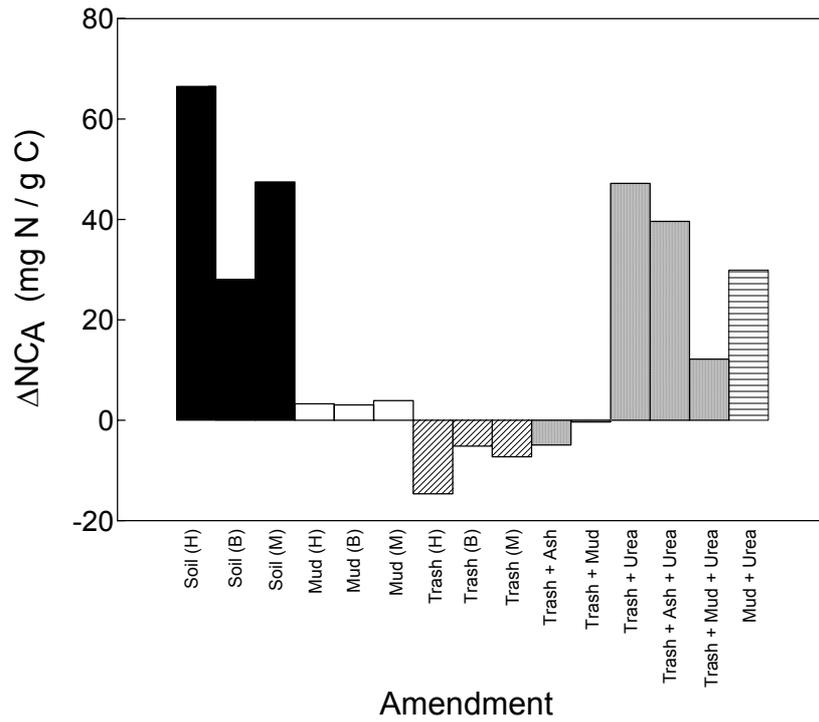


Figure 71 Apparent net N mineralisation or immobilisation per unit of C mineralisation (ΔCN_A)

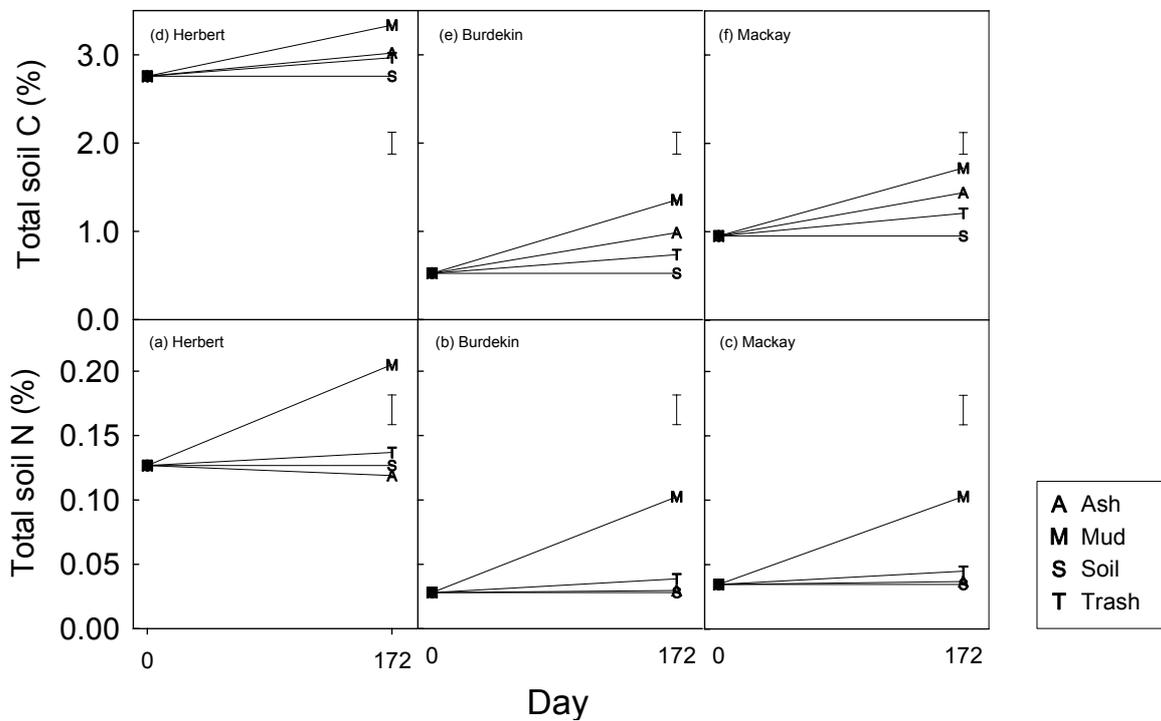


Figure 72 Effects of amendments on total soil C and N. Bars are LSD (P = 0.05) from soil type x amendment interaction

Trash ¹⁵N

Recovery of trash-¹⁵N at the end of the experiment averaged 87%. Recovery from trash alone was slightly less in the Herbert soil than in the other soils (Figure 73). In the Mackay soils, ¹⁵N recovery was generally not affected by the amendments, except for a small reduction where trash, mill mud and urea were applied together (Figure 73).

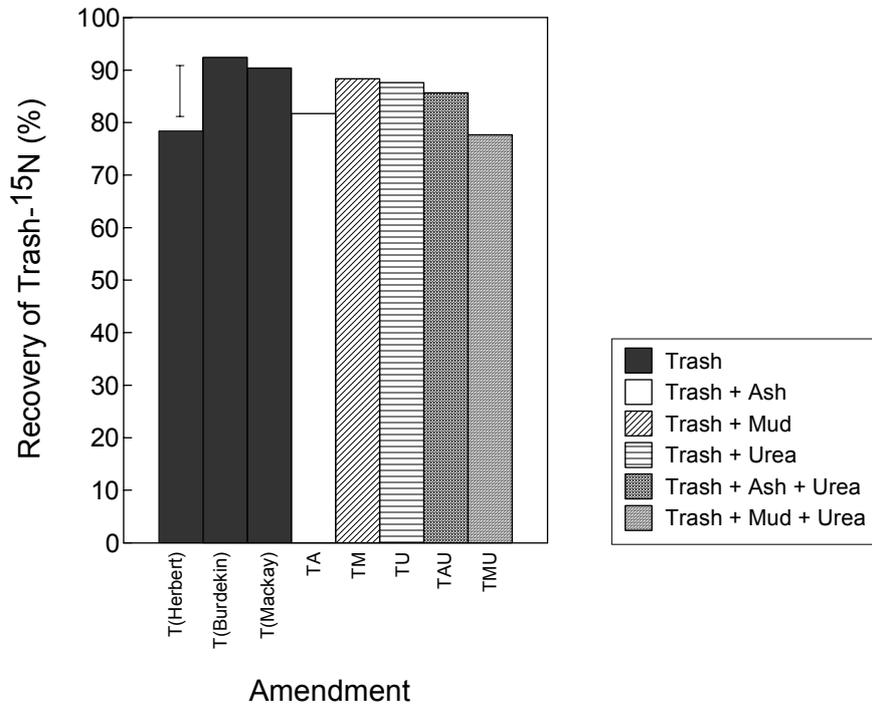


Figure 73 Recovery of ¹⁵N from labelled trash. Bars are LSD (P = 0.05) from one-way ANOVA)

8.3.3 Discussion

The soil amendments investigated in the study - mill mud, mill ash, urea, and sugarcane trash - all had different effects on soil C and N mineralisation.

Mill ash was essentially inert in terms of C mineralisation. This is not unexpected, considering that it had been burnt in the sugar mill furnace, where any labile organic matter would have been destroyed. For this reason too, ash had almost no effect on N mineralisation. In the Burdekin soil, ash incorporation greatly improved soil friability, and this may have resulted in the very small increase in ΔN_A observed.

Mill mud and trash both provided large amounts of decomposable C. On the basis of DM or total C content, mill mud contributed less C (113 mg /g DM, and 420 mg /g C applied) than trash (219 mg /g DM, and 550 mg /g C applied). However, the rates at which these amendments are usually applied in the field means that a typical application of mill mud (150 t/ha moist, 40 t DM/ha) will add about twice as much labile C to the soil as a typical trash blanket (10 t DM/ha).

Mill mud caused net mineralisation of soil N, whereas trash caused net N immobilisation. This is in keeping with the original C:N ratios of these amendments. Net mineralisation of N from mill mud was consistent in the three soils, and was approximately the same as net N mineralisation from the unamended Burdekin and Mackay soils, and rather less than that from the Herbert soil. Trash immobilised approximately the same amount of N as was mineralised from the unamended soils, immobilisation being greater in the Herbert than the other soils.

The largest ΔN_A was produced by urea (almost 10 times greater than the ΔN_A from mill mud). Of the 90 mg N /pot applied, 74 mg N /pot was recovered in the urea-only treatments at the end of the experiment. This discrepancy may have resulted from microbial immobilisation, physical stabilisation, or denitrification of N. As expected, urea had no effect on C mineralisation as the latter was not limited by N availability.

Mixtures of trash, ash, and urea showed C and N mineralisation trends that agreed with their effects when applied alone. With mixtures containing mill mud, however, ΔC_A and ΔN_A were reduced in comparison with mineralisation from individual amendments (Figure 70). The relatively reduced ΔC_A in the mixtures may have been due to a reduction in the efficiency of C use by the microbial biomass, because of over-supply of labile C. The decrease in ΔN_A in mixtures, however, is likely to be due to increased loss of N by denitrification in the presence of mill mud, also caused by the large supply of labile C. That ΔN_C was also smaller in mixtures containing mill mud than would be expected from the ΔN_C of individual amendments substantiates the suggestion of denitrification (Figure 71). Leaching was not a consideration in this experiment, as the pots were sealed at the base.

Although the results presented in Section 7 indicated that there was enhanced decomposition of trash due to the surface application of millmud, results of the micro investigations indicate that ΔC_A was smaller when mud and trash decomposed together than when they decomposed alone. As individual millmud only treatment was not included in the investigations reported in Section 7, a comparison of this kind was not possible. However, it is valid to conclude that both investigations show that decomposition associated with the combined trash/millmud treatment was better than trash alone. If millmud applied to the surface of trash causes better decomposition than trash and millmud individually, then a possible explanation is that the soil microbial population preferentially decomposed trash-C in the presence of mill mud. Despite the positive effects of application of millmud to trash, further work will be required to adequately understand the interactions. Differences in methodology between the experiments reported in this section and those described in Section 7 will need to be taken into account. Here, the mill mud was from a different source, was dried, was applied at a smaller rate, and was incorporated into the soil. Importantly the scale of investigation used here was much smaller than that used in Section 7 and did not link to a field investigation

The ΔN_A from urea alone *minus* the ΔN_A from trash alone was considerably greater than the ΔN_A from the trash + urea treatment (Figure 69). This suggests that: (i) N immobilisation was promoted in the presence of urea; (ii) denitrification was promoted in the presence of trash; or (iii) both processes were promoted simultaneously.

The ΔN_A from mill mud alone represented 3.2% of the total N applied. This is an unexpectedly small proportion for almost 6 months of decomposition of a labile substance with such a low C:N ratio (22). It contrasts markedly with the approximately 20% of total N considered to be mineralisable in the first year following mill mud application to cane crops in the field (Calcino 1994). In the present experiment, there is a strong possibility that significant denitrification also occurred when mill mud was applied alone, and that ΔN_A was therefore underestimated. It is possible, however, that the low ΔN_A was partly due to the particular mill mud used in this experiment.

The apparent denitrification losses in this experiment have important implications for future studies involving mill mud and / urea and trash. The risk of N loss could be controlled by including a shorter sampling period than was used here, or by periodic leaching the soils with water to avoid accumulation of inorganic N, or by incubating at a lower soil water content. The concomitant measurement of C mineralisation can provide a check on the reliability of apparent N mineralisation measurements. The potential for mineralisation and loss of N from mill mud clearly requires further study.

Recovery of trash- ^{15}N at the end of the experiment (mean 87%) indicated that ash, mud and urea had little effect on N loss from trash. Any differences in microbially-immobilised ^{15}N could not, however, be determined. The significant increase in total soil N caused by mill mud application suggests that mud will stimulate net N mineralisation into the longer term.

8.4 Conclusions

Green trash initially caused more C mineralisation than senesced trash, but whether the trash was green or senesced made no difference to DM or C loss at the end of the experiment. Net N mineralisation occurred in green trash and net N immobilisation occurred in senesced trash. Oven-drying the fresh trash had no effect on DM loss, C or N mineralisation. Ground trash temporarily mineralised more C than chopped trash, but there was no difference in final DM loss, C or N mineralisation between ground and chopped trash. There was a strong interaction between trash placement and soil water content: in dry soil, DM loss and C mineralisation were greater when trash was incorporated with the soil; in wet soil, DM loss, C mineralisation, and apparent N mineralisation were greater when trash was surface-applied. There was evidence of N loss through denitrification when trash was incorporated into wet soil.

In such controlled-environment decomposition experiments, it is most important to control the proportion of green and senesced trash, the placement of the trash, and the soil water content. Measurement of decomposition as DM loss is not recommended where the trash fragment size has been greatly reduced.

Mill ash had a negligible effect on soil C and N mineralisation. Trash greatly increased C mineralisation and resulted in strong net N immobilisation. Mill mud greatly increased C mineralisation, and resulted in a small amount of net N mineralisation. Urea provided ten times more mineral N than mill mud, and did not affect C mineralisation. These trends were the same in three contrasting soil types. Mixtures containing combinations of ash, trash, and urea showed C and N mineralisation that agreed with mineralisation patterns

when these amendments were applied alone. Mixtures containing mill mud, however, apparently mineralised slightly less C and much less N than was mineralised from the individual amendments. Significant loss of N through denitrification was suspected in all treatments containing mill mud.

9.0 OUTPUTS

This project has had a number of useful outputs. These include:

- Data and knowledge relating to the effects of trash on soil acidity;
- Information regarding the influence of Ca-containing amendments on trash decomposition;
- Additional information on the management of sodic and acid soils;
- Further knowledge regarding N transformations with trash retention;
- Information about nutrient availability due to trash/Ca-containing amendment interactions;
- Recommendations for growers to maximise the interaction between trash and Ca-containing amendments in acid and sodic soils.
- Advice on liming strategies for trash retention systems.

Following the previous order of reporting, important developments, results and conclusions are summarised below.

9.1 Effects of trash retention on soil acidity

Here we determined:

- a. The net proton (H^+) input resulting from long-term trash retention from existing field trials.
 - b. Influence of changed management of sugarcane on chemical properties.
 - c. Effect of water-soluble organic components leached from plant material.
- Trash retention in the long-term trash retention trials resulted in increases in soil organic carbon at each of the sites, particularly in the surface soil layer. The extent of this increase appears to be a function of period under GCTB.
 - The increase in organic carbon, particularly at the Tully site (the longest under GCTB), has in turn resulted in a shift in the buffering capacity and an increase in proton retention. The difference in net proton input (H^+) between the GCTB and burnt systems was calculated as 4.14 kmol H^+ /ha over an 8-year period.
 - Ash alkalinity has been used extensively to describe the acid-base balance in foods, grains and fodder crops and is a measure of the organic anion production that is produced internally by the plant in order to balance excess cation uptake. It is the decomposition of these anions of organic origin that influence the proton pool. Ash alkalinity of the trash material can be expressed as $CaCO_3$ equivalents if it is assumed that 1 mol H^+ requires 50 g $CaCO_3$ to be neutralised.
 - The net amount of alkalinity in the form of $CaCO_3$ equivalents associated with the trash ranges from 10.23 to 14.6 kg $CaCO_3$ /tonne dry matter. These rates of alkalinity being returned on an annual basis are small compared to the litters collected from

some plant species (i.e. *Melia azedarach* 126 kg CaCO₃/tonne dry matter) and therefore do not contribute significantly to acid neutralisation associated with nitrogen mineralisation.

- It is suggested that in the case of the burnt treatment, the increase in pH is associated with the release of this alkalinity. The potential generation of acid through the mineralisation of nitrogen contained in the trash is negligible since this source is lost on combustion.
- Although the retention of trash resulted in an increase in the charge characteristics through the accretion of soil organic matter, the release of alkalinity from this process was not sufficient to neutralise the acidity generated by the mineralisation of nitrogen in the trash.
- Organic carbon and charge characteristics of soil will increase with time as a consequence of trash retention. This positive consequence on soil chemical properties is important when assessing the long-term impact of changed management systems. The use of charge fingerprints is useful as an assessment of this.
- Charge fingerprints were determined on samples collected from the 0-2 cm depth interval from the Tully site. The difference in charge fingerprinting between the burnt and trash retention system and convergence of the charge curves at approximately pH 5.5 is of significant agronomic importance as it clearly shows that by increasing the pH to this value one would eliminate any potential problems associated with acidic cations (ie Al³⁺). At pH 5.5 the amount of charge generated by the trash retained and burnt treatments was 4.6 and 3.3 cmol_c/kg, respectively. This suggests that the positive impact of trash retention has been the increased ability of this surface horizon to retain nutrients.
- Whilst it is evident that acidic cations have been retained on the exchange complex under the trash retention treatment, the application of lime would effectively neutralise this potential constraint and result in a substantial capacity to retain nutrients. It is suggested that this technique of charge fingerprinting is an effective means of assessing the impact of changed management on the soil resource base.

9.2 Trash decomposition, soil sodicity and the efficacy of calcium-containing amendments and trash for ameliorating sodic soils

Here we determined:

- a. Trash decomposition and soil sodicity.
 - b. Effects of lime, gypsum, trash and their interactions on sodic soils and sugarcane growth.
 - c. Structural stability of sodic soils as influenced by gypsum/molasses.
 - d. Structural stability of sodic soils as influenced by molasses and lime.
 - e. Effect of trash management on susceptibility of soils to becoming remaining sodic.
- Decomposition of sugarcane trash was unaffected by high levels of soil sodicity (SEP 18 – 27) in a light- and a medium-textured soil, and was reduced by 18% in a heavier-textured soil. The latter effect was attributed to impaired soil aeration due to a combination of high sodicity, high clay content, high pH and relatively low organic matter content. Decomposition was unaffected by milder sodicity (ESP 8-12) across all soils

- It is unlikely that soil sodicity will be an important limit to trash decomposition in sugarcane fields where viable crops can be grown.
- In the uncropped pot trial (using sodic soils from the Burdekin and Proserpine), lime and molasses, singly and in combination, increased electrical conductivity (EC) and reduced exchangeable sodium percentage (ESP). For EC, the effects of the two treatments were additive. In the Burdekin soil, lime increased pH, molasses decreased pH, and in combination there was a slight increase in pH. In the Proserpine soil, lime increased pH substantially, whereas molasses and molasses + lime increased pH slightly.
- In both soils, lime and molasses, singly or in combination, decreased the amount of spontaneously and mechanically dispersible clay. The effect was most marked for spontaneously dispersible clay in the Burdekin soil and least marked for mechanically dispersible clay in the Proserpine soil.
- In the Burdekin soil, wet aggregate stability was increased by lime and molasses, with their combined effect being additive. In the Proserpine soil, wet aggregate stability was increased by molasses but was not affected by lime.
- Lime and molasses, singly and in combination, show promise as ameliorants for improving the physical properties of sodic soils.
- The Gapon constant was significantly influenced by trash management in the Tully trial (the longest of the three GCTB trials considered), and by cropping history in the South Australian trial. The difference between Gapon constant between treatments at each site were consistent with differences in organic carbon concentrations and associated differences in cation exchange capacity. (Explanation: Cation exchange selectivity refers to the fact that some cations are held more strongly than others. For example, the cations Ca^{2+} and Mg^{2+} , which have two charges (divalent), are held more strongly than Na^+ and K^+ , which have one charge (monovalent). However, the difference between binding strength of divalent and monovalent cations depends on the nature of the cation exchange sites. Exchange sites on organic matter generally have higher relative selectivity for divalent cations than exchange sites on clays. Therefore, soils with high organic matter content are generally less susceptible to becoming sodic than soils with low organic matter content. The relative preference of a material for these two cations is described by the Gapon constant, K_G . The higher the value of K_G , the lower the relative preference of the material for Ca^{2+} relative to Na^+ . A soil with a high value of K_G is more likely to become sodic under the influence of saline water, and more difficult to ameliorate, or make less sodic, than a soil with a low value of K_G . Therefore, management practices that lower K_G values, are beneficial from a sodicity management point of view. Cation exchange sites in soil occur on clay or organic matter. Exchange sites on organic matter tend to have lower values of K_G than exchange sites on clay particles, so increasing soil organic matter by trash retention may lower the overall K_G of the soil).

9.3 Effect of calcium-based amendments on trash decomposition

Here we determined:

- a. The effect of various calcium amendments on trash decomposition and soil amelioration.
- b. The combined effect of calcium-based amendments and trash retention on nutrient availability.

- Calcium-based amendments, such as lime and millmud, have the potential to accelerate the decomposition of sugarcane trash.
- This decomposition appears to be most effective when the amendments, particularly lime, millmud and mill ash, are applied to the surface of the trash layer.
- Lime had a significant effect on trash decomposition, but the extent of decomposition was dependent on the method of lime application and soil type. Spreading lime on the surface of the trash layer enhanced decomposition.
- As expected, incorporating lime into the soil beneath trash increased soil pH. Spreading lime onto the trash layer also resulted in marked increases in soil pH values in the surface soil (0-10 mm), but although there was evidence of small pH increases after seven months in the 10-50 mm soil layer (from the application of lime onto the surface of the trash), these differences were not significant. However, the substantially lower exchangeable Al values associated with the 10-50 mm soil layer at 4 and 7 months was evidence that some amelioration had occurred due to the application of lime to the surface of the trash. It appeared that the interaction between lime application and trash may possibly have important implications for the amelioration of sub-soil acidity with minimum tillage systems.
- Data from two glasshouse pot trials conducted on five different soil types in different environments (drier subtropical versus moist tropical) indicated that application of certain amendments onto the surface of sugarcane trash had a significant effect on trash decomposition. Relative to unamended trash, the application of filtermud most successfully enhanced the rate of trash decomposition. Of the other amendments considered, significant differences were also applicable when lime (3 t/ha), potassium hydroxide (1.7 t/ha) and urea (160 kg/ha) were applied to the surface of the trash.
- Soil type, treatment and depth below the trash layer had significant interactive effects on soil pH. As expected, the surface applied lime increased soil pH across the five soil types. This was most pronounced in the surface layers and generally declined with depth. The presence of a trash blanket (in the absence of any amendments) had variable effects on soil pH. In some soils the effect on soil pH was either similar or less than that of surface applied lime (no trash). However, in the case of the kurosol, lime application to the trash blanket resulted in higher pH values in the subsurface layers than when lime was applied in the absence of trash. This has important implications for certain acid soils where amelioration of subsurface layers can be enhanced by this interaction of lime and trash, possibly due to more soluble organo-calcium complexes.
- Mill mud application to the surface of the trash layer resulted in some amelioration of the subsurface layers, although the effect was not as pronounced as that with lime.
- The simulated environmental differences between the drier subtropics and moist tropics (drier and cooler versus wetter and warmer) were reflected in the rate of decomposition of the trash (the amount of residual trash expressed as a percentage of the initial amount of trash in each pot). Significantly less trash remained at Macknade (higher temperature and moisture) than at Indooroopilly after the 7-month period in all treatments (including unamended trash) except millmud. As a result, we conclude that millmud appears to consistently enhance trash decomposition over a fairly broad range of moisture and temperature conditions.

- Enhanced trash decomposition that occurred due to millmud and millmud/mill ash, and lime applied to trash in pots under semi-controlled conditions was also found to be applicable in the field.
- Although millmud and mill ash applications to trash resulted in significantly higher amounts of N, P and K removed by the harvested sugarcane biomass, these elevated values could either have been the result of the nutrients within the amendments themselves, from the trash, or a combination of both.
- Millmud or millmud/mill ash mixtures both have positive effects on trash decomposition and ameliorative effects.
- Environmental effects (water, temperature and soil properties) are considered important factors in determining nutrient availability in the amendment/trash systems.

9.4 Trash decomposition and C and N relationships

Here we determined:

- a. Trash decomposition in controlled conditions.
 - b. Trash decomposition and effects of trash, millmud, mill ash and urea on soil C and N.
- When measuring decomposition of crop residues under glasshouse or laboratory conditions, it is desirable to approximate the conditions under which the residues decompose in the field. Close simulation of field conditions is, however, difficult to achieve because experimentation on a small scale (10-cm-diameter pots) with limited resources requires the researcher to make compromises.
 - In small pot experiments, factors such as the size to which the residue is cut; whether the residue is fresh or dried; the placement of the residue in or on the soil; and the soil water content vary from the field situation. The extent to which these factors affect decomposition of sugarcane trash is unknown, but studies with residues of other crops suggest that all these factors may be important.
 - To help choose experimental conditions for future decomposition studies, the effects of these factors on decomposition of sugarcane trash were determined.
 - Trash may be safely oven-dried and roughly ground before application to the soil, which greatly simplifies handling and subsampling of the trash.
 - The strong interaction between trash placement and soil water content means that these factors must be carefully controlled in decomposition experiments.
 - Measurement of trash decomposition as DM remaining at the end of the experiment is not an adequate method for small-scale studies where the trash fragment size had been greatly reduced.
 - Measurement of decomposition as CO₂ loss is a more reliable and sensitive method, and allows decomposition to be measured non-destructively over time (it is also much easier).
 - Although it has been suggested, by growers and scientists, that trash decomposition may be increased by incorporating the trash into the soil by cultivation, the results of the small-scale pot experiments indicate that incorporation of trash may possibly hinder decomposition in situations where the soil is wet for long periods.
 - Application of straight mill ash does not appear to improve soil C or N availability (though it may improve the availability of some other nutrients and have non-nutritional benefits).

- When millmud and/or urea are applied in the presence of trash, N availability may initially be reduced due to microbial immobilisation.

9.5 General

The advantages of retaining trash in the field can be further enhanced by the application of ameliorants to the surface of the trash blanket. Generally, the amendments, such as lime, millmud and millmud/mill ash mixtures applied to the surface of a trash layer have positive results for sugarcane cane production. Lime applied to trash has the ability to counter acidity generated by the mineralisation of N from the organic matter. Sources of organic matter (trash, millmud, molasses, etc), whether present individually or in combination with each other, have the ability to ameliorate sodic soil conditions. The potential of ameliorants such as lime and millmud to accelerate trash decomposition when applied to the surface of the trash layer has important implications for improving the return of organic matter to the soil, for ameliorating of soil acidity, and ultimately improving nutrient-use efficiency. Rates of up to 5 t/ha of lime (as determined by soil tests) and between 100 and 200 t/ha of wet millmud or millmud/mill ash mixtures applied to the surface of trash layers during the crop cycle would be appropriate to enable growers to reap the benefit of using Ca-based amendments in combination with trash retention. This practice and these rates are achievable in the sugar industry, as lime and millmud are often applied to sugarcane prior to planting.

Although some technology transfer has occurred in terms of papers, presentations and posters, further extension of the concept is proposed. In particular, a paper entitled 'Cane trash and Ca-based soil amendments: A combination too good to miss' will be presented at the 2004 ASSCT conference. Further extension will make use of grower meetings and workshops.

10.0 EXPECTED OUTCOMES

- A clearer understanding of the effect of GCTB on soil organic matter and its ability to increase the buffer capacity of soil, particularly where trash retention has been practiced for an extended period.
- Greater awareness of the H⁺ inputs from trash and the influence of trash retention systems on soil chemical properties.
- Increased use of GCTB on sodic soils, with the knowledge that trash decomposition is generally not affected by soil sodicity.
- Increased use of millmud and mill ash as amendments applied to the surface of the trash blanket for amelioration of acid soils.
- Increased use of millmud and millash for improving nutrient-use efficiency.
- Increased use of lime applied to the surface of the trash layer for ameliorating acid soil conditions.
- Use of Ca-based amendments to accelerate the decomposition of trash in circumstances where a trash blanket has detrimental effects on the growing crop (cool, wet climates, water-logged conditions, tropical areas, etc).

- Better use of combinations of lime, gypsum and molasses in combination with trash blanketing for ameliorating the physical effects of sodic soils.
- Minimal wastage of nutrients for sugar industry by products and reduced off-farm impacts.
- Greater awareness amongst growers, extension staff and agribusiness representatives for the need to use any synergies of GCTB and amendments usually applied to cane and/or those available in the industry as by-products.
- Greater adoption of practices that lead to best practice management on-farm.

11.0 FUTURE RESEARCH NEEDS AND RECOMMENDATIONS

A large amount of research and development that has been undertaken in relation to GCTB, especially within CRC Sugar projects, and has provided much information about the advantages and limitations of trash retention. This project has added to the pool of knowledge by evaluating the effects of applying commonly used ameliorants to the surface of trash layers. Although additional information is still required to validate the results from the small-scale pot experiments (Section 8) under field conditions, much of the information is ready for extension to the industry.

Some technology transfer has occurred (Section 12) and in particular, a paper titled 'Cane trash and Ca-based soil amendments: A combination too good to miss' (BL Schroeder and AW Wood) is planned for presentation at the conference of the Australian Society of Sugar Cane Technologists in Brisbane in 2004. Apart from this, a concerted effort is needed to communicate many of the outputs listed in Section 9 to the industry via a range of extension methods. A booklet or information sheet summarising the important aspects would be an advantage.

To enable more widespread use of the positive interaction between trash retention and Ca-based amendments, research is needed to enable millmud and mill ash to be more widely used in canegrowing districts. At present, the relatively high water content makes transport of these products over longer distances uneconomical.

Further field trials are needed to validate whether the range of outputs are appropriate in districts other than the dry subtropics (Bundaberg) and moist tropics (Herbert). Although the investigations relating to sodic soils and the C and N relationships were conducted on soils from the Burdekin, Mackay and Proserpine, larger pot experiments and/or field trials are needed to evaluate the effect of other environments (especially rainfall and temperature) on trash decomposition in the presences of lime, millmud, mill ash, etc.

An important by-product, dunder, was not included in this investigation. In the event of increased ethanol production in the Australian sugar industry, the implications of applying combinations of Ca-based products and dunder to trash layers will be important.

Research is also needed to evaluate the availability of micro-nutrients within the amended trash systems. This could have important implications for soils with marginal to low trace element reserves.

12.0 PUBLICATIONS ARISING FROM THE PROJECT

Presentations and published papers

- Marschner B & Noble AD. 2000. Chemical and biological processes leading to the neutralisation of soil acidity after incubation with different litter materials. *Journal of Soil Biology and Biochemistry* 32, 805-813.
- Schroeder BL & Aitken RL. 1998. Effect of lime on green cane trash decomposition. National Soil Acidification Conference, 15-17 July 1998, Noosa, Australia.
- Schroeder BL & Wood AW. 2004. Cane trash and Ca-based soil amendments: A combination too good to miss. Proceedings of the Australian Society of Sugar Cane Technologists in prep.
- Suriadi A, Murray RS, Grant CD & Nelson PN. 2002. Structural stability of sodic soils in sugarcane production as influenced by gypsum and molasses. *Australian Journal of Experimental Agriculture* 42, 315-322.
- Suriadi A, Murray RS, Grant CD & Nelson PN. 2002. Influence of crop management and water quality on the exchange selectivity of soils. ASSSI Future Soils Conference, 2-6 December 2002, Perth.
- Suriadi A, Murray RS, Grant CG & Nelson PN. 2000. Structural stability of sodic soils as influenced by gypsum and molasses. Sodicity Conference, 28 February – 1 March 2000, Tatura.
- Suriadi A, Murray RS, Grant CD & Nelson PN. in prep. Sodium-calcium exchange selectivity of soils under different trash management practices.
- Wood AW, Schroeder BL & Aitken RL. 2001. The effect of various calcium amendments applied to sugarcane trash. 10th Australian Agronomy Conference, Hobart, Tasmania, 28 January – 1 February 2001.

Poster papers

- Noble AD & Berthelsen S. 2000. A new approach to understanding cation chemistry. Presented at CRC Sugar Annual Review and Planning Meeting, Townsville, 6 – 8 March 2000.
- Schroeder BL & Aitken RL. 1999. Effect of lime on green cane trash decomposition. Presented at CRC Sugar Annual Review and Planning Meeting, Townsville, 23 - 25 February 1999.
- Schroeder BL, Wood AW & Aitken RL. 2000. Assessing the effect of various Ca-amendments on the decomposition of trash. Presented at CRC Sugar Annual Review and Planning Meeting, Townsville, 6 - 8 March 2000.

13.0 ACKNOWLEDGEMENTS

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14.0 REFERENCES

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