

SHORT AND LONG TERM IMPACTS OF GREEN CANE TRASH BLANKETING ON SOIL FERTILITY

FINAL REPORT

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Research Organisations:

CSIRO Sustainable Ecosystems
Level 3, Queensland Bioscience Precinct
306 Carmody Road
St Lucia Qld 4067

BSES Limited
PMB 4 Bundaberg DC
Bundaberg Qld 4670

Principal Investigators:

Dr Peter Thorburn
☎ 07-3214 2316
Fax 07-3214 2308
Mob. 0417-073 173
E-mail: Peter.Thorburn@csiro.au

Dr Graham Kingston
☎ 07-4132 5200
Fax 07-4132 5253
Mob. 0419 661305
E-mail: GKingston@bses.org.au



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PROJECT STAFF AND ACKNOWLEDGEMENTS

Project staff

Principal Investigators

CSIRO Sustainable Ecosystems

Dr Brian Keating (to 1997)

Dr Peter Thorburn (from 1997)

BSES Ltd.

Dr Peter Larsen (to 1997)

Mr Ross Mitchell (1998-2000)

Dr Graham Kingston (from 2001)

Research staff

CSIRO Sustainable Ecosystems

Dr Keith Weier

Mr Charlie McEwan

Mr Jody Biggs

Ms Heidi Horan

Ms Kerry Collins

BSES Ltd.

Mr John Jackson

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1. EXECUTIVE SUMMARY

In the last three decades, there has been a widespread adoption of green cane harvesting-trash blanketing (GCTB) in the Australian sugarcane industry, generally motivated by the practical advantages (e.g. weed control, easier harvesting in wet times, etc.) associated with the GCTB farming system. While many studies have been conducted on the impact of trash blanketing on issues such as yield, soil erosion and soil physical properties, surprisingly little is known about the fate of nutrients in trash (whether burnt or retained), the decomposition of trash blankets, or impacts of GCTB on soil fertility and hence fertiliser management. This project aimed to provide information on these issues and so determine if canegrowers need to change their nutrient management strategies when they switch from burnt cane harvesting to a GCTB system.

Principal findings from this project are:

- Burning trash reduces trash dry matter by 70 % (pre-harvest burn) to 95 % (pre- and post-harvest burn), and the loss of nutrients when trash is burnt is strongly related to the loss of dry matter. Thus GCTB can substantially increase organic matter and nutrient retention.
- Rainfall will quickly leach some nutrients, especially potassium (K), from freshly harvested trash into the soil.
- Trash decomposes slower than expected from its biochemical composition, with up to 10% of trash deposited on the soil after harvest still present one year later. However, the decomposition of trash can be accurately predicted across soils types and climates from sugarcane-specific residue decomposition models.
- Trash blanketing, for up to 17 years, increases soil organic carbon (C) and nitrogen (N) concentrations in the top 50-100 mm of soil, but has negligible impact on concentrations of other nutrients in the soil.
- The presence of trash blankets increases denitrification by 20 % on waterlogged, heavy soils in the period immediately following applications of N fertiliser. Denitrification in this situation could be accurately modelled, and this modelling provided the first test of the denitrification sub-model in The Agricultural Productions Systems Simulator (APSIM).
- K from trash is plant-available, so fertiliser and trash K should have similar value as nutrients. However, there was not a significant yield response to K from trash in either pot or field experiments, possibly due to variability in the results. Thus it is not clear that recommendations of K management in trash-blanketed crops should differ from those for burnt crops.
- Simulation studies of N dynamics in GCTB systems suggest that:
 - There may be a negative, short-term impact of trash blanketing on sugarcane yields for at least 5 years after initiating GCTB. This is due to the immobilisation of N by the decomposing trash. During this time it is important that N applications not be reduced below those used when trash was burnt. Following that time;
 - N fertiliser applications to **ratoon** crops in GCTB systems should be maintained at rates appropriate for burnt systems, despite N in trash being recycled in the GCTB

system. The additional N from trash is immobilised by decomposing trash blankets or lost to the environment.

- In **plant** crops, N fertiliser applications in GCTB systems could be reduced to half that recommended for burnt systems.
- Average environmental losses of N, from both denitrification and leaching, are likely to be greater from GCTB systems at all rates of N fertiliser (i.e. less than, equal to or greater than optimal rates) so particular caution should be taken to avoid over-application of N in GCTB systems.

During the project, information was disseminated through the industry in numerous presentations given at field days, shed meetings, Mill Supplier Committee meetings, CRC-Sugar meetings (at which BSES extension officers were present), ASSCT, etc. In addition, a series of workshops was run in collaboration with CRC-Sugar on the impact of trash blanketing on soil fertility and fertiliser management. The workshops were aimed at BSES, productivity board and fertiliser company advisers, and conducted using a participatory approach to information exchange, based on adult learning principles.

2. BACKGROUND

In the last three decades, there has been a widespread adoption of green cane harvesting-trash blanketing (GCTB) in the Australian sugarcane industry. Much of the motivation for this change has been associated with practical advantages associated with GCTB in the farming system, namely easier weed control, improved in-field trafficability, easier harvesting in wet weather and avoidance of sugar loss from burnt cane that cannot be harvested during wet weather (Wood, 1991). As a result, many studies have been conducted on the impact of trash blanketing on issues such as yield (Page *et al.*, 1986; Wood 1986, 1991; McMahon and Ham, 1996; Kingston *et al.*, 1998; Chapman *et al.*, 2001), soil erosion (Prove *et al.*, 1995; Glanville *et al.*, 1997) and soil physical properties (Braunack and Ainslie, 2001). However, despite its widespread uptake by the industry, outside the Herbert region (Wood, 1986, 1991; Sutton *et al.*, 1996) surprisingly little is known about the fate of nutrients in trash when burnt or retained, the biochemical composition of trash and decomposition of trash blankets, or impact of GCTB on soil fertility and how this will affect sustainable crop production, particularly in the long-term.

Crop residue management is widely recognised as playing an important role in many agricultural production systems because of its diverse effects on soil physical, chemical and biological properties (Kumar and Goh, 2000). These changes to the soil properties may affect the soil environment in a way that results in an alteration of nutrient transformations and efficiency of use (Power and Doran, 1988). The degree to which these changes occur has been widely studied across various environments and in a plethora of cropping systems. The results of these studies generally demonstrate beneficial effects from retention of crop residues, such as increased soil organic matter (SOM), enhanced soil structure, improvements in soil nutrients, and influences on soil moisture and temperature regimes (Kumar and Goh, 2000; Schomberg *et al.*, 1994; Lal, 1995). While many benefits of crop residue retention have been reported, the degree to which these occur and the time frame in which they are observed varies widely throughout the literature.

SOM, commonly measured as soil organic carbon (C), has an important function in crop production due to its significant role in soil nutrient provision, its cation exchange and water holding capacities (Schomberg *et al.*, 1994) and its role in aggregate stability, which affects water infiltration, etc (Bell *et al.*, 1999). Numerous researchers have recorded an increase in SOM following crop residue retention. Black (1973) demonstrated that after four years of crop-fallow cycles in a wheat production system in Montana, USA, SOM increased significantly to a depth of 30 cm, proportionate to the quantity of wheat straw incorporated. Maskina *et al.* (1993) also reported a significant increase in SOM to 30 cm after the addition of 6 t/ha of corn residues for five years in Nebraska, USA, while Hooker *et al.* (1982) found a difference to only 15 cm after nine years of stubble retention from irrigated wheat and sorghum crops in Kansas, USA. Long-term additions of organic matter (35 t/ha/yr of manure) at Rothamsted increased total soil C by 2% (0-23 cm), but it took approximately 70 years for soil C to reach equilibrium (Johnston, 1986). Whereas at another site in England, Woburn, it was found that similar additions of manure did not prevent the continual decline in soil C, a drop of 0.8%, in a soil cropped over 100 years (Johnston, 1986). In a long-term sugar cane GCTB experiment in South Africa, soil C (0-10 cm) increased by 0.8% after 58 years where residue was retained (van Antwerpen and Meyer, 1998; Graham *et al.*, 2000). The effect of crop residues on SOM content has been shown to be highly related to the amount of residue applied and only weakly related to the type of residue (Rasmussen and Collins, 1991).

The majority of the research into crop residue retention and its effect on soil nutrient status demonstrates that phosphorus (P) tends to accumulate near the soil surface when residues are retained (e.g. Geiger *et al.*, 1992; Prasad *et al.*, 1999; Singh and Singh, 1995; Black, 1973; Juo and Lal, 1979; Larson *et al.*, 1972). Generally it is considered that residues containing less than 0.2-0.3% total P can result in net immobilisation of P, while net mineralisation will occur at higher P contents. In regards to sugar cane trash retention, Wood (1986) and Graham *et al.* (2000) both found that trash treatments resulted in higher available P compared to the burnt treatments. While Ball-Coelho *et al.* (1993) found that total P concentrations were greater in a post-harvest burn treatment than in trash retention plots, six months after residue addition. These higher soil P concentrations do not necessarily result in greater P availability to crops. Many studies found no significant treatment effect by residue retention on soil extractable P (Bationo *et al.*, 1993; Standley *et al.*, 1990; Hooker *et al.*, 1982; Thompson, 1992; Djokoto and Stephens, 1961; Lal 1998).

Numerous studies have shown that, like soil C, soil nitrogen (N) levels are usually enhanced by crop residue retention (Power and Doran, 1988). In general, the returning of residues with an N content greater than approximately 1.5% (or a C:N ratio of ≤ 25) usually enhances soil N availability, whereas returning residues with less N (higher C:N ratio) reduces N availability in the first few years after addition through immobilisation of residue N by the SOM (Power and Legg, 1978). Generally these significant increases in soil N levels have been reported in the upper soil depths only (to 15 cm), with increases not occurring at lower depths (Hooker *et al.*, 1982; Dalal, 1989). Research into the effects of sugar cane residue retention reflect this trend, finding that GCTB results in an increase in total N compared to the burning of cane trash but frequently only in the upper (0-10 cm) soil layers (Sutton *et al.*, 1996; Wood, 1991; Wood, 1986; Graham *et al.*, 2000).

Generally, other soil nutrients are also reported to be enhanced by crop residue retention but again the level of accumulation and the depth to which it occurs can vary widely.

Historically, fertiliser recommendations in the Australian sugar industry have been based on burning cane prior to harvesting as this was the dominant method of trash management when the experiments on which recommendations are based were conducted. The differences in nutrient availability that are expected to occur between farming systems in which crop residues are burnt or retained will have important implications for nutrient management in the sugar industry. Consequently, growers could be either under- or over-fertilising, depending on the relative availability of nutrients under the different production systems. This situation is especially important for N and potassium (K), because these nutrients are required in large amounts to maximise sugarcane yields, and hence the profitability of sugarcane production. Unlike K however, N losses to the environment can be considerable and so it is particularly necessary to avoid over-application of N.

It is difficult to predict the impact of trash blanketing on cane production and N fertiliser management. The studies described above concentrated mainly on measuring only one of the possible sinks of N in a trash blanketed system, i.e. "storage" of N in soil organic matter. Apart from this, other potential fates of N include (1) increased environmental losses of N by leaching or denitrification, or (2) increased N concentrations in cane and, hence, a net removal of N from the site if trash blanketing makes soils "N-rich".

The N cycle in sugarcane production systems is complex in comparison to other cropping systems. Although both sugarcane and cereal cropping systems require N for vegetative phases, cereals transport N to a grain 'sink' as protein that is accumulated with maturation. In sugarcane, maturation is delayed and sugar content reduced if excess N continues to be accumulated in the crop up until harvest (Wiedenfeld, 1995; Muchow *et al.*, 1996). The

length of sugarcane crops (1-2 yr) and the infrequency of fallows in the plant-ratoon crop cycle increases the need for year-round management of N. However, these longer time frames make it more difficult to generalise about N supply from SOM. These characteristics of the N cycle in sugarcane production systems suggest that a sugarcane cropping system model with crop and soil N capabilities able to respond to management practices and climate would be a valuable tool in examining/deriving improved N management strategies.

Since crop residues significantly affect the soil hydrological and N cycles, models that aim to predict the dynamics of the N cycle and crop growth under variable residue management practices also need to accurately predict residue dynamics. The Agricultural Productions Systems Simulator (APSIM; McCown *et al.*, 1996) is one such model. It describes the dynamics of crop growth (for various crops), soil water, soil N and C, and plant residues as a function of climate, cropping history (e.g. crop type, sowing date) and soil management (e.g. tillage, fertiliser application). APSIM's linking of crop growth with soil water and N/C dynamics makes it particularly suited for extrapolating the results of agronomic experiments to different soil, management and/or climatic conditions (McCown *et al.*, 1996; Keating *et al.*, 1997; Probert *et al.*, 1995, 1998). Despite its wide application for this purpose, to date some APSIM modules have only been tested in grain and/or legume systems in semiarid, rainfed areas (Carberry *et al.*, 1996; Probert *et al.*, 1995, 1998) and so could benefit from testing in sugarcane production systems before their application in those systems.

This project aimed to identify the appropriate action to be undertaken by growers to adjust their nutrient management strategies, in accordance with the difference in nutrient availability, when they switch to green cane trash blanketing. This will lead to more efficient nutrient use within the sugar industry and potentially reduce nutrient losses to the wider environment.

3. OBJECTIVES

The objective of this project was to determine if canegrowers need to change their nutrient management strategies when they switch from burnt cane harvesting to green cane harvesting. To establish this, the immediate, short-term and long-term impacts of trash blanketing on the dynamics of essential plant nutrients and soil organic matter were investigated. This broad objective can be broken down into nine more specific objectives. These objectives and the sections in which they are addressed are:

1. Determine the quantity of plant nutrients in trash that will be either lost from, or returned to sugarcane soils by different cane harvesting / trash management systems (Section 4).
2. Determine the short-term dynamics of nutrient release from green cane trash blankets (Section 5).
3. Quantify trash decomposition across the broad environmental conditions of the sugar industry (Sections 6 and 7).
4. Determine the extent to which denitrification is promoted under green cane trash blanketing (Section 8).
5. Analyse soils from existing long-term comparisons of trash management practices to identify key impacts on soil chemical properties and soil organic matter quality (Section 9).
6. Identify the effects of trash blankets on the dynamics of nutrient stratification and release in the medium-term (Section 11).
7. Capture process understanding in quantitative model modules and evaluate the ability of a cropping systems model (APSIM) to predict soil organic matter content, soil nitrogen content and residual trash levels across a broad environmental spectrum (Sections 7, 8, and 10).
8. Interpret experimental studies and use models and long-term climate data to evaluate the long-term impacts of green cane trash blanketing on soil N fertility (Section 12).
9. With key extension officers and industry groups, develop a strategy to raise industry's awareness and understanding of the findings of this project (Section 13).

Much of the work undertaken in this project was closely linked to activities within the CRC for Sustainable Sugar Production.

These objectives were successfully met.

4. FATE OF NUTRIENTS DURING BURNING

Introduction

Retention of sugarcane trash on the soil surface has the potential to increase the fertility of soils. The magnitude of this increase is limited by the quantity of organic matter and nutrients returned in the crop residues that would have been lost by burning. Thus knowledge of amounts of dry matter (DM) and nutrient in trash, and the proportion of these remaining on the soil surface after trash burning is essential for understanding changes in soil fertility and crop fertiliser requirements after adopting GCTB.

The proportion of biomass retained as ash after it is burnt can vary greatly among plants and plant parts (Fearnside *et al.*, 1999), indicating that specific studies may be required to investigate biomass loss when sugarcane residues are burnt. The proportion of potentially useful crop nutrients retained after burning crop residues can also be expected to vary. The N in crop residues is largely contained within the organic fraction and is conventionally considered to be volatile in biomass fires (Raison *et al.*, 1985). As might be expected for this, N loss has been found closely related to total DM losses from cereal straw (Biederbeck *et al.*, 1980). Cereal straw P (Biederbeck *et al.*, 1980) and calcium (Ca) and magnesium (Mg) in Eucalypt forests (Raison *et al.*, 1985) were largely retained when the biomass was burnt. P, Ca, Mg and K have been highly concentrated (10-50 fold) in the ash after Eucalypt forest fires, compared to the unburnt biomass (Raison *et al.*, 1985). Loss of this fine ash could therefore reduce nutrient retention following fires. Recoveries have been linearly related to DM recovery (Raison *et al.*, 1985).

While there have been studies conducted on the fate of DM and nutrient when plant residues of cereals or Eucalypts are burnt, similar information does not exist for sugarcane. Thus experiments were conducted to quantify the proportion and quantity of DM and nutrients returned to the immediate area after burning sugarcane crops. A subsequent objective was to identify relationships between DM and nutrient recovery.

Methodology

Field burning

Seven experiments were conducted in 1995 and 1996. Six were near Mackay, in central Queensland, Australia, the remaining site in Tully, north Queensland.

All experiments consisted of four plots, each 7.5m wide and 20m long. In each experiment, three treatments were applied sequentially (repeated measures) to four separate plots:

1. **Residues unburnt.** Twenty cane stalks were hand-harvested from each plot. The tops were collected as the shoot material above the recognisably millable cane. All other attached leaf material was separated and defined as the trash. Combining the top and trash residues is subsequently referred to as total residues.
2. **Residues burnt pre-harvest only.** Four steel trays were placed at ground level towards the centre of each plot. A conventional sugarcane fire was then set through the entire plot. After the fire had finished, ash and residue material was collected from the trays. Crop residues still connected to the cane was removed from 20 hand-harvested stalks, as for Treatment 1. The material collected from the trays and that removed from the stalks was

combined as the total residue after treatment 2. The cane was then mechanically harvested and the fallen residues collected on further steel trays placed at ground level.

- Residues burnt pre- and post-harvest.** As the sugarcane was harvested, tops and trash fell to the ground. This residue was raked into windrows. Four steel trays (1m²) were placed separately under the windrows. These residues were burnt and DM and nutrient contents of the material on each tray were measured. The mass and nutrient (N, Ca, Mg, P, K and sulphur (S)) contents of residues collected after each treatment were determined.

All results were expressed on a dry weight basis.

Laboratory ignition

Six separate samples of sugarcane tops and trash, each weighing about 5-10kg (fresh), were collected in Mackay immediately before harvest of commercial crops in 1999. Each sample was then cut into lengths of ca. 5cm and mixed prior to sub-sampling. Two replicate subsamples (~ 25g) were dried at 80°C and the remaining mass measured to allow calculation of DM content.

A further four sub-samples of tops or trash (separate) were then placed in pre-heated incineration ovens at 600 or 900°C for 5 minutes, with two replicates per temperature. This resulted in a combination of 2 replicates x 2 plant parts (trash and tops) x 2 temperatures. The mass loss on ignition of each sample was then calculated and expressed on a dry weight basis.

Results

Field burning

The return of DM and all nutrients was reduced significantly by the pre-harvest burning and ranged from 12% for Ca to 30% for K (Figure 1). Additional post-harvest burning only significantly reduced the recovery of DM and N (7 and 4%, respectively: Figure 1). Compared to trash retention burning cane residues can therefore reduce the return of nutrients by ca. 55 kg/ha N, 35 kg/ha Ca, 20 kg/ha Mg, 7 kg/ha P, 50 kg/ha K, 7 kg/ha S and 10t/ha DM.

The burning of the sugarcane reduced the return of DM and all nutrients collected separately as trash and tops (Figure 2). Only 11-23% of DM and nutrients in trash, and 41-77% of the tops, were recovered after the pre-harvest fire.

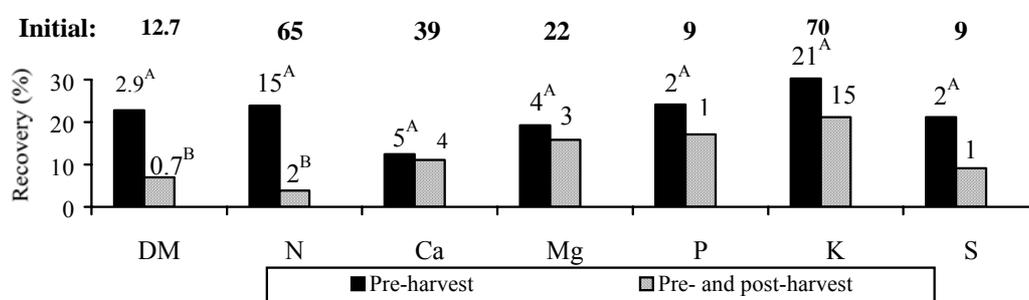


Figure 1. Dry matter (numeric data in t/ha) and nutrient (numeric data in kg/ha) recovery after pre-harvest and pre- and post-harvest burning sugarcane residues (mechanically harvested).

^A: Recovery of DM or nutrients after burning cane before harvest was significantly different (5% level) to the initial (unburnt) content.

^B: Recovery of DM or nutrients after burning cane before and after harvest was significantly different (5%) to the

recovery after a pre-harvest burn only.

The proportion of N, Ca, Mg, P and S recovered after the pre-harvest burn of the residues, was linearly related ($p < 0.05$, $r^2 = 52-78\%$) to the loss of DM (Figure 3a). Linear relationships ($p < 0.05$, $r^2 = 31-62\%$) were again observed for these nutrients collected in the tops and trash after the pre-harvest fire (Figure 3b). K recovery was consistently not significantly related to DM recovery ($p > 0.05$, $r^2 = 8-20\%$).

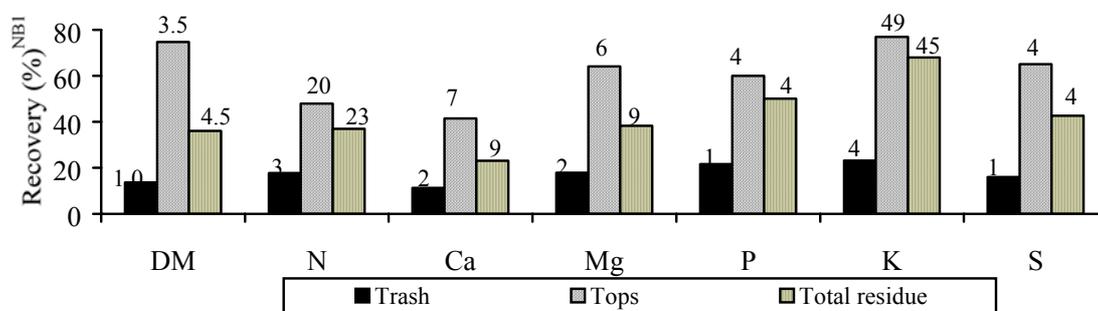


Figure 2. Dry matter (numeric data in t/ha) and nutrient (numeric data in kg/ha) recovery from sugarcane trash and tops collected by hand after burning

^{NB1}: Each recovery is a percentage of the individual residue components initial content, or their combined content for the tops and trash together.

^{NB2}: Recovery after burning cane was significantly different (5% level) to the initial (unburnt) content for all nutrients and DM.

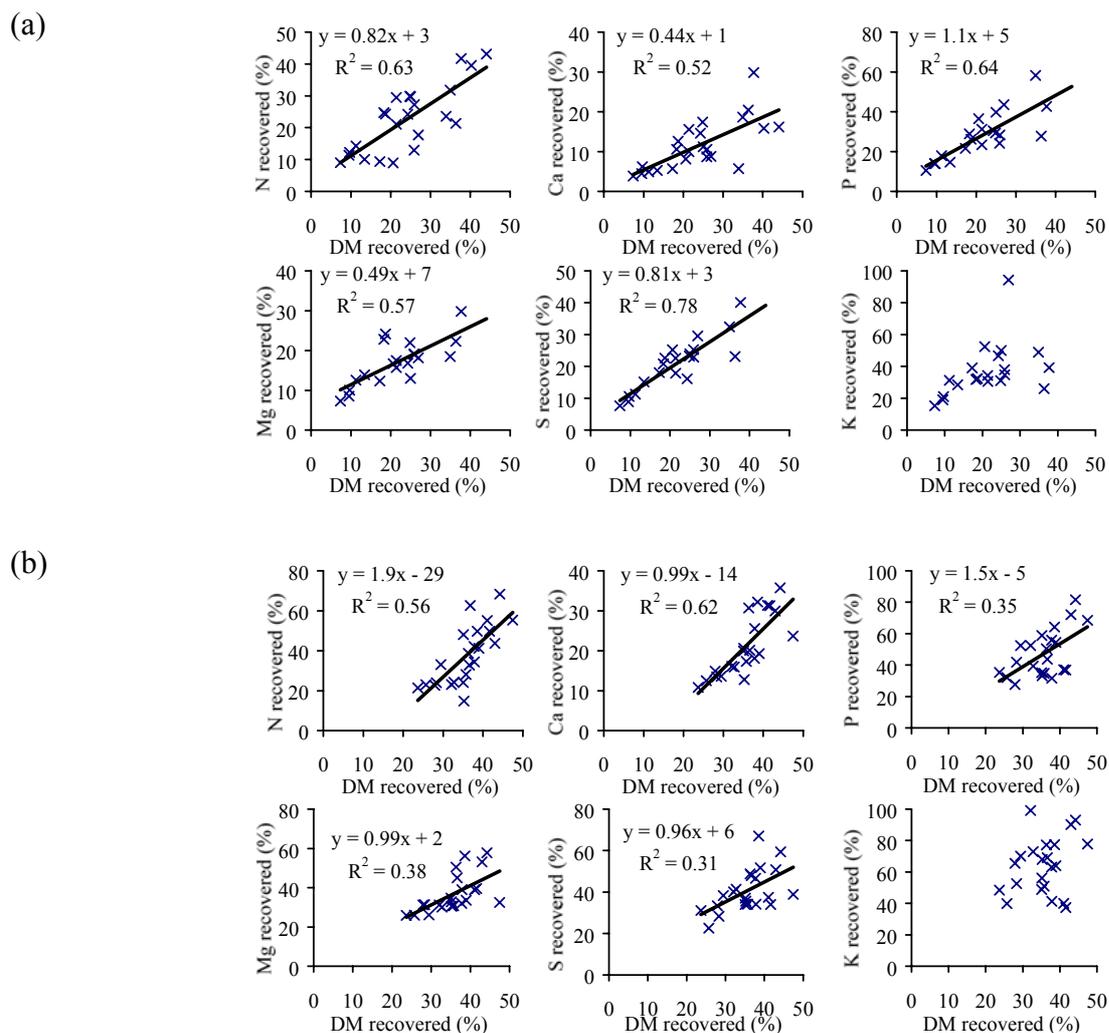
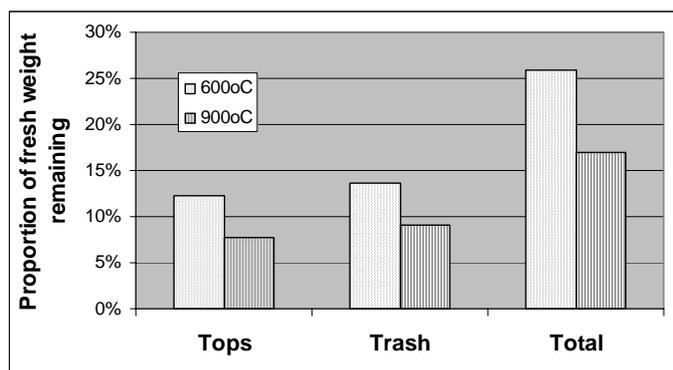


Figure 3. Relationships between nutrient and dry matter (DM) recovery after: (a) pre-harvest burn of sugarcane (mechanical harvest), and (b) burnt trash and top harvest (total) residues collected after hand harvest.

Laboratory ignition

There was a significant reduction in DM recovery when the furnace temperature was increased from 600 to 900°C. The DM recovery from furnace ignited trash and tops (17-26%) were similar (Figure 4) and comparable with recoveries when these residues were burnt in a standing crop and mechanically harvested (Figure 1). The DM recovery from the furnace ignited trash (9-14%: Figure 4) was also comparable to trash recovered from the field (14%: Figure 1). The DM recovery from the furnace-ignited tops (8-12%: Figure 4) was considerably less than tops recovered as tops after the pre-harvest fire in the field (75%: Figure 2).

Figure 4. Dry matter recovery after burning isolated sugarcane trash and tops in a furnace.



Discussion

Changing to a system that retains sugarcane residues, from one that burns them, should increase the return of macronutrients. This may subsequently increase their concentration and availability in the soil. The return of all nutrients however, was largely dependent on the component of the sugarcane residue burnt (trash or tops) and the methodology adopted to collect them (removal after hand or via mechanical harvest). Recovery of DM and nutrients after mechanically harvesting the burnt standing cane crop (5-30%) was significantly less than the total recovered from the burnt and hand collected trash and tops (23-68%). Nutrient recovery can vary after fires of differing flaming characteristics (Susott *et al.*, 1991), or environmental variables, particularly wind velocity (Biederbeck *et al.*, 1980), and the physical characteristics of the material burnt (Fearnside *et al.*, 1999). All these characteristics were inherently the same within these experiments, as the burnt residues were collected from the same plots after the same fires. The only variable was the method of harvesting, suggesting a considerable loss of crop residues from the immediate area during mechanical harvesting. Hand harvest and removal of crop residues provides valuable information about the relative nutrient recoveries and is a useful check for field techniques. We suggest that the recoveries after burning the standing crop are the most representative of those likely in the field.

There were generally strong linear relationships between DM and nutrient recovery, except for K. Previous work has shown strong relationships between DM and nutrient recovery, including K (Raison *et al.*, 1985). We found that K recovery was consistently greater than other nutrients, but a large majority of K was still lost. Other work has found that P was largely recovered, with post-fire recovery of 100% from cereal straw (Biederbeck *et al.*, 1980). Nutrient recoveries tended to be Ca > P > K > N after fires in Eucalypt forests (Raison *et al.*, 1985). Recoveries in our experiments were generally K > P > Mg > S > N > Ca. The recovery of N and particularly Ca, tended to be less than most other nutrients and DM. The small N recoveries are consistent with those from burnt cereal straw (25%), where P recovery was 100% (Biederbeck *et al.*, 1980). This N recovery is consistent with the range we have observed (11-41%). However, within the experimental series reported by Biederbeck *et al.* (1980) a far greater proportion of N was recovered in less windy conditions (83%).

These results support the view that GCTB can increase DM and nutrient retention. The data presented indicates that, under field conditions, gross losses from the immediate area after burning sugarcane trash, can be considerable (N, 77-95%; Ca, 87-89%; and K, 70-73%). The potential increase, however, will be dependent on the longer-term fate of ignition products exported from the immediate area and their net redistribution within sugarcane growing areas. Most redeposition in Florida, USA, occurred 5-8 km downwind from sugarcane fires

(Achteimeier, 1998). We are not aware of any data that can adequately quantify exhaust product movement over a larger scale under Australian conditions.

5. SHORT-TERM NUTRIENT RELEASE FROM TRASH BLANKETS

Introduction

Nutrients in trash (and other crop residues) exist in various chemical forms. Some, such as N, are bound up in plant tissues and are only released as those tissues decompose. Others exist in soluble forms that can be rapidly leached from the trash into soil by rainfall or irrigation. These latter nutrients will be available to the crop in the short-term, whereas those released through the process of trash decomposition will only become available in the medium- to long-term. Understanding the time scales of release of different nutrients from trash is important for developing fertiliser management recommendations for different trash management practices.

This experiment aimed to determine the quantity of N, Ca, Mg, P, K and S leached from trash in the short-term. The leaching was undertaken to be equivalent to the first rainfall event following deposition of trash on the soil.

Methodology

The experiment was conducted in the laboratory, at a constant temperature of 25° C. There were four replicates of each treatment. Trash from Q124 was collected immediately after harvest and air-dried in a glasshouse for 4 weeks prior to the experiment. Trash (at a rate equivalent to 9.2 t ha⁻¹) was placed between two polyethylene disks (in the leaching equipment), wet to saturation and allowed to stand for 1 hour. Five volumes of deionised water, 19.7, 30.7, 42.3, 54.6 and 91.4 mm (equivalent ponded depth), were then applied to the trash. The trash was then collected, oven dried, ground, then analysed for elemental composition (described above). Samples were also taken from the trash prior to leaching, and analysed for elemental composition.

Results

Leaching of water through trash immediately reduced the quantity of Mg, K and S in the trash (Table 1). Potassium decreased exponentially, with approximately 60 % of the K in the trash being leached with 20 mm of water, and greater than 95 % of the K in the trash being removed with greater than 55 mm of water. The quantity of Mg and S leached from the trash remained constant after the first leaching event.

Nitrogen was not removed from trash collected immediately after harvest and prior to decomposition, despite 91 mm of water being leached through the trash.

Phosphorus and calcium contamination of the water used to leach the trash resulted in an increase in the quantity of P and Ca measured in the trash. Unfortunately no corrections could be applied to this data due to a failure to measure the quantity of P and Ca in leachates and hence derive a mass balance.

Table 1. The mean content (and standard error of the mean) of N, Ca, Mg, P, K and S in 9.2 t DM/ha of trash prior to leaching, and the effect of leaching differing volumes of water through trash on this nutrient content.

Treatment	N	Ca	Mg	P	K	S
Initial (kg/ha)	29.0	18.6	10.8	4.07	24.5	4.99
	(1.52) ^A	(0.686)	(0.420)	(0.230)	(3.12)	(0.287)
Initial (% of initial)	100	100	100	100	100	100
19.7 mm leachate	99.8	103	84.7	132	38.5	64.8
30.7 mm leachate	103	106	76.9	146	17.6	56.8
42.3 mm leachate	107	116	81.8	133	6.10	61.8
54.6 mm leachate	103	112	76.5	126	2.53	61.4
91.4 mm leachate	101	107	84.7	112	2.15	63.5
HSD	NS ^B	NS	12.2	27.8	27.4	16.4

^ANumbers in brackets are standard errors of the mean.

^BNo significant difference between treatments ($P > 0.05$).

Discussion

Numerous authors have reported that K, Mg and S are rapidly leached from freshly harvested organic materials (e.g. Parker, 1983; Kalburtzi *et al.*, 1990; Spain and Hodgen, 1994; Tietema and Wessel, 1994). The results from this experiment concur with the conclusion of these authors that inorganic constituents of K in particular and then Mg and S in sugarcane trash are readily available for leaching. Thus rainfall events will quickly move these nutrients from freshly harvested trash in the cane field into the soil, where it should be more accessible to the plants roots. The rapid movement of K from the trash and the large quantities involved may have significant impacts on grower fertiliser programs for this nutrient.

The present experiment suggests that N movement from trash is dependent on the transformation of organic N into inorganic forms before it can be leached from the trash. This result supports the significance of research programs examining the decomposition of trash and the subsequent effects on N in the soil.

Phosphorus has been reported to rapidly leach from organic materials (Spain and Hodgen, 1994; Tietema and Wessel, 1994). In the present experiment contamination precluded any conclusions being made on the immediate availability of P to the soil due to leaching events. However, given the small quantity involved the value may be insignificant in the overall fertilising program of farmers.

6. QUALITY OF RESIDUES FROM SUGARCANE CROPS

Introduction

Decomposition of crop residues is dependent on their biochemical properties. Commonly, this is expressed as the amount of C, and proportions of various forms of C, such as lignin, phenols, celluloses, soluble C, etc. (Heal *et al.*, 1997). The concentration of N, and the amount of N relative to total C (i.e., the C/N ratio) or lignin (i.e., the lignin/N ratio) are also commonly used to characterise the quality of trash. These latter parameters, particularly C/N ratio, are of particular note as they are commonly used to model decomposition of plant residues (Probert *et al.*, 1998; Heal *et al.*, 1997).

Characterising crop residues is further complicated by the presence of different plant parts, such as leaf and stem, in residue. A perennial crop such as sugarcane will also have both dead leaves and tops in residue blankets on the soil surface. Different cultivars may also differ in the quality of the residues. While there has been some limited chemical analysis of trash blankets (Spain and Hodgen, 1994) and it is known that N concentrations differ in different parts of the plant and through time (Wood *et al.*, 1996), there is little information of different forms of C, and how these differ in different parts of the plant and across different varieties.

This study aimed to characterise the biochemical quality of trash, and the different constituents of trash (live and dead leaves, and stem) for different cultivars and across different environments. It also aimed to determine the relationships between C/N ratio and lignin/N ratio of trash, since these are such common parameters describing trash in cropping systems models.

Methodology

Site description

Samples were taken from five long-term residue management field experiments located at three sites, Tully, Mackay and Maclean (Table 2). The sites were distributed along approximately 1,500 km of the coast of north-eastern Australia, from near the southern extreme of the Australian sugarcane industry (Maclean) to one of the wettest tropical regions of the industry (Tully). Annual rainfall and average temperature varied considerably between the sites. There were two residue management experiments located at each of the Mackay and Maclean (BS168) sites. The experiments at these sites differed in their plant/harvest date; one termed “early harvest” and the other “late harvest”. The early harvest experiments were planted/harvested in July each year, early in the harvesting season, while the late harvest experiments were planted/harvested late in the harvesting season, November, each year. In each experiment, sugarcane had been planted at a row spacing of 1.5 m, annual N fertiliser applications were approximately 180-200 kg ha⁻¹ and there were 3 or 4 replicates.

Sampling

Two groups of samples were taken. The first group was an inter-site comparison of residue quality. Samples were taken of residue that had been prepared for field measurements of residue decomposition in one of the treatments at each of the five experiments. In each experiment, a large quantity of residue (e.g. > 15 kg DM) was collected from each replicate plot 1-12 days after mechanical harvest of the crop and mixed to increase uniformity in the distribution of green and dead leaves. A sub-sample of this residue was taken for chemical

analysis. The remainder was returned to the field and spread over eight quadrats in each replicate plot. A sample of the residue was taken destructively from one quadrat each 1-2 months, but only data from the 1st and second sampling times are reported here. The variety Q124 was sampled from all experiments in this group, allowing site-to-site and temporal variations in quality to be characterised.

The second group of samples came from different varieties of sugarcane growing in one of the experiments, the Mackay late harvest experiment. In this group, separate samples of green leaves, dead leaves and cane were obtained from six different varieties; Q124, Q135, Q136, Q138, Q159 and H56-752. The samples were obtained by hand harvesting 20 stalks of cane randomly chosen from each plot, the day prior to mechanical harvesting. The stalks separated into the different components by hand and a sub-sample of approximately 1 kg weight randomly chosen for subsequent chemical analysis.

Table 2. Details of sites sampled.

Site	Mackay (M)	Tully (T)	Maclean (W)
Latitude, Longitude	21.10 S, 149.07 E	17.56 S, 145.56 E	29.50 S, 153.20 E
Average rainfall (mm/yr)	1,670	3,480	1,160
Average temperature, maximum / minimum (°C):			
Summer	31 / 23	32 / 23	29 / 20
Winter	22 / 12	25 / 14	20 / 7
Soil texture	Loam	Clay loam	Loam / Clay
Total N 0-0.1 m depth (%)*	0.08	0.12	0.21
Total C 0-0.1 m depth (%)*	1.32	1.33	2.53
Year treatments imposed	1993	1991	1996
Replicates	4	3	4
Mean residue C:N	Late harvest – 96:1 Early harvest – 76:1	117:1	Late harvest – 78:1 Early harvest – 69:1
Date decomposition measurements commenced	Late harvest – Nov. 1996 Early harvest – Jul. 1997	Nov. 1996	Late harvest – Nov. 1996 Early harvest – Jul 1997

* Mean across the site.

All samples were analysed for total C and N and lignin. For the inter-site group there was enough sample for determination of soluble C at all sites at the second sampling time (referred to as Soluble C(s2)), but only enough at the two early harvested sites for determination at the first sampling time referred to as Soluble C(s1)). Soluble N was also determined on the samples from the inter-site group. Total C and N were determined by dry combustion and lignin by the acid-detergent fibre method of Goering and Van Soest (1970). Soluble C was extracted by shaking 1 g (dry matter weight) of residue with 40 ml of deionised water (25°C) for 1 hour. The C concentration of the extract was determined by the method of Owczkin *et al.* (1995) and results expressed relative to total C content of the residue.

Results

The quality of the residue varied between the five sites, resulting in a wide range (69-117) of residue C/N values across the sites (Table 3). The early harvested experiments at Mackay (ME) and Harwood (WE) had lower residue C/N and lignin than the late harvested experiments at those locations, and C/N tended to increase with decreasing latitude. Soluble C at the second sampling time (Soluble C(s2), Table 3) also tended to be highest at these sites. As expected, there was a marked decrease in soluble C between the first and second sampling times at the ME and WE sites. Rain between the two sampling times would have removed the soluble C, as it is rapidly leached from trash. Lignin and C concentrations were correlated ($P < 0.06$), as were the ratios lignin/N and C/N ($P < 0.001$).

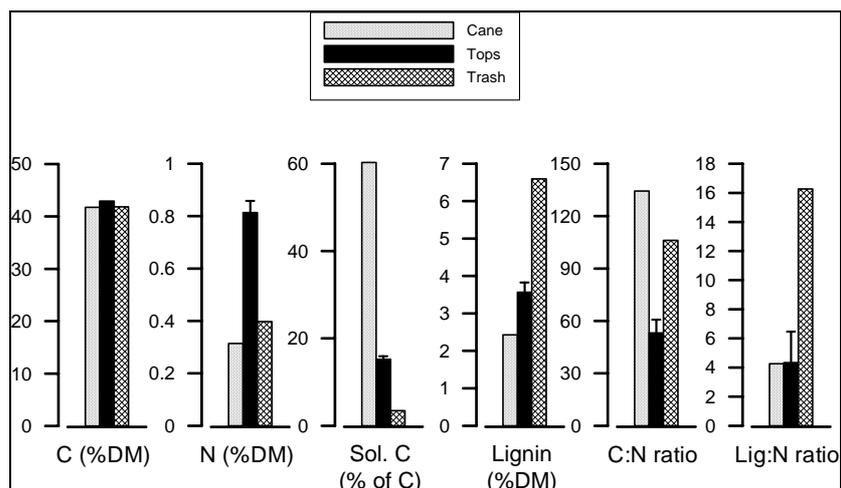
There were no significant differences in N concentration, C/N or soluble C concentration between different sugarcane varieties grown at the ML experiment. However, there were significant ($P < 0.001$) differences between varieties and significant ($P < 0.05$) variety by plant part interactions in lignin and lignin/N. The mean lignin concentration for each variety ranged from 3.6 to 4.8 %, with a corresponding range in lignin/N of 6.6 to 10.4.

As expected, there were significant differences ($P < 0.01$) in all chemical characteristics between the different plant parts (Figure 5). The difference in lignin concentrations and lignin/N ratios between plant parts was much greater than between varieties. The ratios lignin/N and C/N were significantly correlated ($P < 0.001$) whether data were pooled across all plant parts or analyses conducted on the separate plant parts. Lignin and C concentrations were not significantly correlated ($P < 0.05$), when analyses were conducted on the separate plant parts.

Table 3. Quality of residue sampled from the five sites at the beginning of the decomposition experiments, except for soluble C(s2) which is for samples from the second sampling time. Standard errors are given in parenthesis.

Analysis	Site				
	ME	ML	T	WE	WL
Lignin	5.0 (0.22)	6.4 (0.28)	5.0 (0.17)	3.9 (0.14)	5.3 (0.09)
Soluble C(s1)	3.5 (0.19)			4.7 (0.25)	
Soluble C(s2)	1.7	0.9	0.7	1.6	1.0
Total N	0.60 (0.023)	0.46 (0.018)	0.38 (0.032)	0.67(0.057)	0.57 (0.030)
C/N	76 (2.7)	96 (4.3)	117 (11.6)	69 (5.5)	78 (4.1)
Lignin/N	8.2 (0.41)	14.1 (1.07)	13.2 (1.35)	6.0 (0.67)	9.3 (0.45)

Figure 5. Average (across varieties) proportions of N and C quality in the different parts of above ground biomass.



Discussion

The main determinant of biochemical quality of residues left on the ground following harvest of a sugarcane crop will be the ratios of the different components of the crop in the residue. The variety of sugarcane grown will have a negligible impact. Cane, tops and dead leaves vary significantly in all biochemical attributes except C concentration (Figure 5). The ratio of these components in residues left after sugarcane is harvested will be markedly affected by whether cane is burnt prior to harvest and on the “quality” of the harvest operation. Burning prior to harvesting will greatly reduce the amount of dead leaves in the residue, and so increase the average N concentration of the residue. Poor quality harvesting will increase the amount of cane in the residue, increasing the average C/N ratio. Thus a broad range of residue quality is possible at any site and will need to be accounted for in modelling residue decomposition.

Another factor that may influence the biochemical characteristics of sugarcane residue is the variation that occurs in cane and tops N concentrations with crop age (i.e., the time since the crop was planted or last harvested) and N supply to the plant. N concentrations in tops decrease with crop age and N concentrations in tops and cane will greatly increase in crops to which too much N fertiliser has been applied (Muchow *et al.*, 1996; Wood *et al.*, 1996). Thus a crop which is harvested early (e.g., 10 months old) and over fertilised with N will have considerably higher average N concentrations in residue than crops of more average age and recommended N management. These management effects will enhance the site specific nature of sugarcane residue quality discussed above.

In models of crop residue dynamics, residue quality is generally described by C/N or lignin/N (Heal *et al.*, 1997). Given that C/N and lignin/N ratios are well correlated in sugarcane, the latter parameter is unlikely to provide improved predictions of residue decomposition for sugarcane. That has significance for modelling decomposition of sugarcane residues because determination of lignin is much more time consuming than N.

7. DECOMPOSITION OF TRASH BLANKETS

Introduction

The amount of crop residues present on the soil surface, and their rate of decomposition significantly affect the soil hydrological and N cycles. The amount of residue on the soil surface is a function of both residue deposition on the surface and subsequent decomposition. The greatest amount of deposition occurs at harvest, although during crop growth senesced leaves can detach from the plant and add to residues on the soil surface. Decomposition is a biological process with bacteria and fungi important decomposers (Wardle and Lavelle, 1997). However, the role of macro-invertebrates is also important in physically transporting fragments of residue beneath the soil surface (Tian, *et al.*, 1997; Wardle and Lavelle, 1997).

There have been numerous studies of crop residue decomposition. However, in the majority of these the residue has been buried in soil, as would occur following ploughing (e.g., Mary *et al.*, 1996). The relatively few studies of decomposition of surface residue have generally been undertaken in areas where retention of crop residues on the soil surface is important for control of erosion, such as in the cereal producing belts of USA and Australia (e.g. Sallaway *et al.*, 1988; Thomas *et al.*, 1990; Steiner *et al.*, 1999; Stott *et al.*, 1990; Stroo *et al.*, 1989). Much of the variation in decomposition (and subsequent mineralisation of N) in different experiments can be explained by environmental factors (i.e. rainfall and temperature), and residue C:N ratio (Mary *et al.*, 1996; Dimes, 1996, Probert *et al.*, 1998; Whitmore and Handayanto, 1997; Steiner *et al.*, 1999; Stott *et al.*, 1990; Stroo *et al.*, 1989). While explaining decomposition in the context of these parameters does not provide a description of the fundamental processes (i.e., the specific soil biological processes) controlling decomposition, it does provide a framework allowing quantification of the process, if even only at a functional level.

There have been few measurements of residue (trash) decomposition reported for sugarcane. Spain and Hodgen (1994) found that the mass of a trash blanket declined by 81 % over approximately one year at a site in the Herbert region. Subsequent to the work reported below, Klok *et al.* (2003) found similar amounts of trash decomposition at sites in the wet tropics. These studies provide an important first step in quantifying trash blanket decomposition. Measurement of environmental factors and trash quality will allow trash decomposition to be compared with decomposition of other crop residues, and the prediction of decomposition under varying management conditions.

Because of the importance of surface residues in soil water and N dynamics in farming systems, models that aim to predict the dynamics of crop growth under variable residue management practices must accurately predict residue dynamics. The Agricultural Productions Systems Simulator (APSIM; McCown *et al.*, 1996) describes the dynamics of crop growth (for various crops), soil water, soil N and C, and plant residues as a function of climate, cropping history (e.g., crop type, sowing date) and soil management (e.g., tillage, fertiliser application). APSIM's linking of crop growth with soil water and N/C dynamics makes it particularly suited for extrapolating the results of agronomic experiments to different soil, management and/or climatic conditions (McCown *et al.*, 1996; Keating *et al.*, 1997; Probert *et al.*, 1995).

Despite its wide application for this purpose, to date some APSIM modules have only been tested in grain and/or legume systems in semiarid, rainfed areas (Carberry *et al.*, 1996;

Probert *et al.*, 1995, 1998) and so could benefit from wider testing. This is particularly so for sugarcane production systems, since the climate, soils and agronomic characteristics (e.g., residue mass, N fertilizer application rates, frequency of fallows) of these systems are markedly different from those where the APSIM-Residue module was developed and most widely used.

As is common in modelling crop residue (and soil organic matter) dynamics (Paustian *et al.*, 1997; Whitmore and Handayanto, 1997), decomposition in APSIM-Residue is described by first order kinetics (see below). The rate coefficient is formulated to account for environmental and 'quality' attributes of the residue, such as some biochemical attributes and physical attributes (like particle size). This approach is used because soil biological processes controlling decomposition are highly complex and incompletely understood. Experience suggests that the rate coefficient is applicable to a particular family of plants (such as cereals; Carberry *et al.*, 1996; Probert *et al.*, 1995, 1998) but may need to be varied for different families of crops or types of plants (Steiner *et al.*, 1999; Snow *et al.*, 1999).

This study aimed to quantify decomposition of trash blankets under a wide range of conditions (e.g., climates and times since the cessation of burning) to confirm how widely applicable the relatively slow decomposition previously observed is. These data will then allow the testing and, if necessary, improvement in modelling decomposition of sugarcane residues with APSIM-Residue.

Methodology

This activity aimed to quantify the breakdown of trash blankets and develop an accurate means of modelling decomposition of sugarcane residues with APSIM-Residue. Predictions of sugarcane residue decomposition were compared with measurements in five field experiments located in a wide range of environments in the Australian sugar industry. A new function describing decomposition rate as a function of residue mass was developed for APSIM-Residue. The value of the parameter in this function, and values of selected constants in other functions in APSIM-Residue were then determined by optimisation against data from two of the experiments. The applicability of the parameter values to other sugarcane systems was then assessed against the independent measurements of decomposition in three other experiments conducted in more temperate and tropical environments than the first two experiments.

Site description

Sugarcane residue decomposition was measured over one year in the five field experiments sampled for the biochemical quality studies described above (Table 2). Each experiment had treatments comparing burning and retention of residues after harvest. The residue management treatments had been imposed for different lengths of time, up to 5 years, before the experiments started. The variety Q124 was common to all experiments, so measurements were made on that variety.

Measurements

Within 12 days of harvesting the crop, eight quadrats, 1.5 m x 0.75 m in area, were established in each replicate plot of the experiments. A known mass of residue was enclosed within each quadrat. Residue in the quadrats was in direct contact with the soil, but enclosed over the top with fine nylon mesh, approximately 20 mm in diameter. The mass of residue in individual quadrats was measured destructively every 1-2 months. At the time when the quadrats were established in the experiments, residue samples were also collected for determination of total C and total N.

Daily rainfall, temperature and solar radiation were measured at each site. Soil moisture was logged hourly with Campbell CR10X dataloggers and CS615 water content reflectometers in one replicate plot at three of the experiments, Mackay late harvest (ML), Mackay early harvest (ME) and Maclean early harvest (WE). Measurements were made over the 0-50 mm soil depth interval, and at 75 mm and 175 mm depth in both the residue retained and residue burnt treatments. Daily average values were calculated from the data.

Modelling

APSIM-Residue

In the APSIM system, plant residue on the soil surface is treated as a component separate from soil organic matter and incorporated residue. C in the surface residues is transferred into the soil organic matter pools of the soil N module (APSIM-SoilN; Probert *et al.*, 1998) by two processes, incorporation (e.g. during tillage) or decomposition. The rate of residue decomposition is controlled by first order kinetics,

$$dR/dt = -kR, \quad (1)$$

where R is the mass of residue per unit area, t is time, and k is the rate coefficient given by,

$$k = D_{\max} \cdot F_{C:N} \cdot F_{\text{temp}} \cdot F_{\text{moist}} \cdot F_{\text{contact}}, \quad (2)$$

where D_{\max} is the maximum (or potential) decomposition rate and $F_{C:N}$, F_{temp} , F_{moist} and F_{contact} are factors, scaled from 0 to 1, accounting for the limitations to decomposition imposed by residue C:N ratio, temperature, moisture and residue-soil contact, respectively. The functions defining the factors (Figure 6) are,

$$F_{C:N} = \exp(0.277(1 - C:N/C:N_{\text{opt}})), \quad C:N > C:N_{\text{opt}}, \quad (3)$$

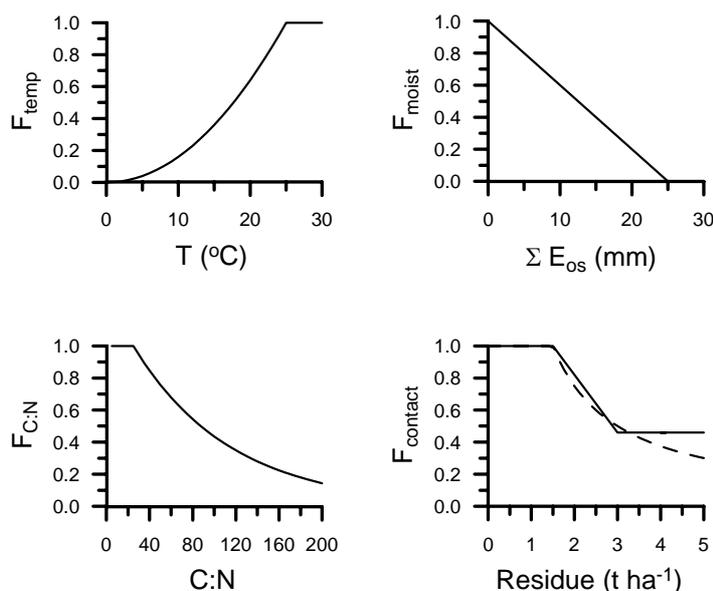
$$F_{\text{temp}} = (T / T_{\text{opt}})^2, \quad T \leq T_{\text{opt}}, \quad (4)$$

$$F_{\text{moist}} = 1 - (\sum E_{\text{os}}) / E_{\text{os,max}}, \quad (\sum E_{\text{os}}) < E_{\text{os,max}}, \quad (5)$$

$$F_{\text{contact}} = 1 - \left(\frac{1 - F_{\text{contact,min}}}{R_{\max} - R_{\min}} \right) (R - R_{\min}), \quad R_{\min} < R \leq R_{\max}, \quad (6)$$

where, T is the average daily air temperature, E_{os} is the potential daily soil evaporation, R_{\min} is the mass of residue above which the rate of decomposition is reduced, R_{\max} is the mass of residue above which the rate of decomposition is independent of mass, and the subscripts 'opt' and 'max' refer to the optimum and maximum parameter values, respectively. For F_{moist} , E_{os} is summed each day until there is more than 4 mm of rainfall and/or irrigation, at which time E_{os} is reset to zero. Outside the limits given in equations 3 to 6, the factor values are set equal to 1 or 0, except for F_{contact} which has a minimum ($F_{\text{contact,min}}$) of 0.46 at $R > R_{\max}$ in the default values. The default values for the other parameters are given in Figure 6. E_{os} is related to potential evaporation calculated from the Priestly-Taylor equation in APSIM's soil water balance model (SOILWAT; Probert *et al.*, 1998).

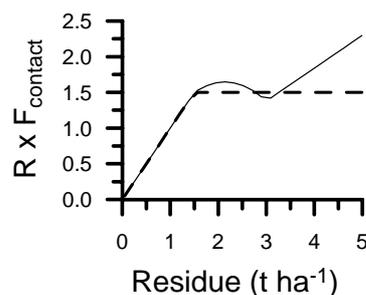
Figure 6. The default functions for the four factors (equations 3-6) that limit residue decomposition in APSIM-Residue due to (a) temperature ($T_{opt} = 25^{\circ}\text{C}$), (b) moisture ($E_{os,max} = 25$ mm), (c) initial residue C:N ratio ($C:N_{opt} = 25$), and (d) residue-soil contact ($R_{min} = 1.5$, $R_{max} = 3 \text{ t ha}^{-1}$ and $F_{contact, min} = 0.46$). The dashed line in (d) shows the new residue-soil contact function (equation 7) with $R_{crit} = 1.5 \text{ t ha}^{-1}$.



A new contact factor

Decomposition rates (i.e., dR/dt) per unit mass of residue are lower with larger initial residue masses (e.g. Steiner *et al.*, 1999; Dimes, 1996; Stott *et al.*, 1990; Stroo *et al.*, 1989), due to the separation of the upper mulch layers from the soil micro-organisms and the generally drier environment in these layers (Christensen, 1986; Dimes, 1996). Thus decomposition of a large mass of residue does not necessarily obey first order kinetics and k should be a function of R . This is achieved through the inclusion of $F_{contact}$ in equation 2. The impact of such a factor on the decomposition rate can be simply assessed from the relationship between R and $R \cdot F_{contact}$ (Figure 7), as, from equations 1 and 2, $R \cdot F_{contact} \propto dR/dt$. For equation 6, the slope of this relationship becomes negative as R increases from R_{min} to R_{max} . It would be expected that the slope should be ≥ 0 at all values of R , and so a different function to describe contact limitations is desirable.

Figure 7. The impact of residue mass on residue decomposition rate (represented by $R \cdot F_{contact}$) resulting from the different contact factor functions, equations 6 (solid line) and 7 (dashed line). Parameter values for equations 6 and 7 are given in Figure 6.



A simple function consistent with these concepts can be developed by conceptualising residue consisting of two layers (Figure 8): layer 1, residue in contact with the soil and decomposing; and layer 2, all the residue above layer 1 which is not decomposing for the reasons given above. Residue decomposed from layer 1 would be “replaced” by residue from layer 2. The mass (thickness) of layer 2 would decrease through time, but the mass of layer 1 would remain constant until there was no residue in layer 2. Thus, the rate of decomposition would be independent of R until sufficient decomposition had occurred that layer 2 had ceased to exist. If the mass of residue in layer 1 is R_{crit} , then an expression for a modified contact factor, $F_{contact, 2}$, is

$$F_{contact, 2} = R_{crit} / R, \quad R > R_{crit} \quad (7a)$$

$$F_{\text{contact}, 2} = 1, \quad R \leq R_{\text{crit}}. \quad (7b)$$

This expression results in a continuous function at $R \geq R_{\text{crit}}$ (Figure 6) and the slope of the relationship between $R \cdot F_{\text{contact}}$ and R is ≥ 0 at all values of R (Figure 7).

It is difficult to assign a value to R_{crit} . Stott *et al.* (1990) and Stroo *et al.* (1989) found that wheat residue decomposed more slowly (per mass of residue) at an initial mass of $\sim 6 \text{ t ha}^{-1}$ than at $\sim 3 \text{ t ha}^{-1}$. In contrast, Dimes (1996) found that residue of a forage legume (*Stylosanthes hamata*) decomposed more slowly at an initial mass of 3 t ha^{-1} than at 1.5 t ha^{-1} . Thus, the value of R_{crit} may be residue specific.

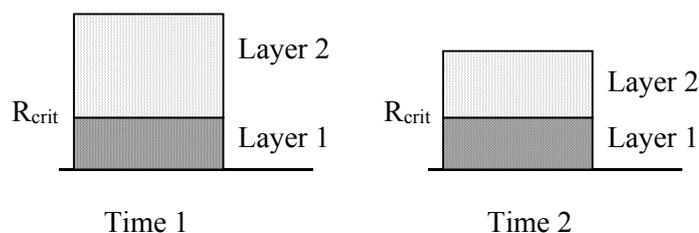


Figure 8. A conceptual diagram of residue on the soil surface showing two layers: layer 1, in contact with the soil and decomposing; and layer 2, the residue above layer 1 that is not decomposing. The thickness of layer 1 remains constant through time while there is residue in layer 2. The thickness of layer 2 decreases through time as residue from that layer “replaces” the residue decomposed from layer 1.

Model parameter optimisation

Apart from the need to derive a value for R_{crit} , it is possible that some of the default parameter values in APSIM-Residue may be inappropriate for sugarcane. Thus, a numerical optimisation was undertaken to determine parameter values that would result in more accurate predictions of sugarcane residue decomposition. For the optimisation, APSIM-Residue was programmed into a spreadsheet based on a daily time step and data required for the model derived from measurements at the experimental sites. The value of $F_{\text{C:N}}$ was determined from measurements of residue C:N ratio (Table 2) when the quadrats were established. The inputs for calculating the daily values of F_{temp} were taken from air temperature measurements at the sites. In the three experiments (ME, ML and WE) where soil moisture was measured, E_{os} data for calculating daily values of F_{moist} (equation 5) were derived from water balance calculations performed on the soil moisture data. In the other two experiments, Maclean late harvest (WL) and Tully (T), E_{os} was estimated as a constant proportion (10 %) of potential evaporation (derived from the weather data). Neither the proportion used, nor the substitution of more complex functions to estimate E_{os} (such as relating E_{os} to rainfall and potential evaporation) had a marked effect on predicted decomposition at these two experiments. Using these input data, daily values of k were calculated from equation 2, and then daily residue mass from equation 1.

Optimal values were determined for R_{crit} , D_{max} and T_{opt} . These parameters were chosen because (1) there was no information available on the value of the newly derived parameter, R_{crit} , (2) previous studies had suggested that the value of D_{max} (Snow *et al.*, 1999) and equivalent parameters in other residue decomposition models (Steiner *et al.*, 1999) may be crop specific, and (3) it was thought the default value of T_{opt} (20°C) may be too low compared with values used in other decomposition models (e.g. 33°C ; Stroo *et al.*, 1989), especially in a tropical system like sugarcane production. The parameters in the function for F_{moist} were not examined because the simplistic estimation of E_{os} in the WL and T experiments did not provide information of sufficient reliability for evaluating the function.

The parameters in the function for $F_{C:N}$ could not be examined because the generally high values of residue C:N ratio in the experiments (≥ 69 , Table 2) relative to $C:N_{opt}$ (25) did not provide a good test for the value of $C:N_{opt}$.

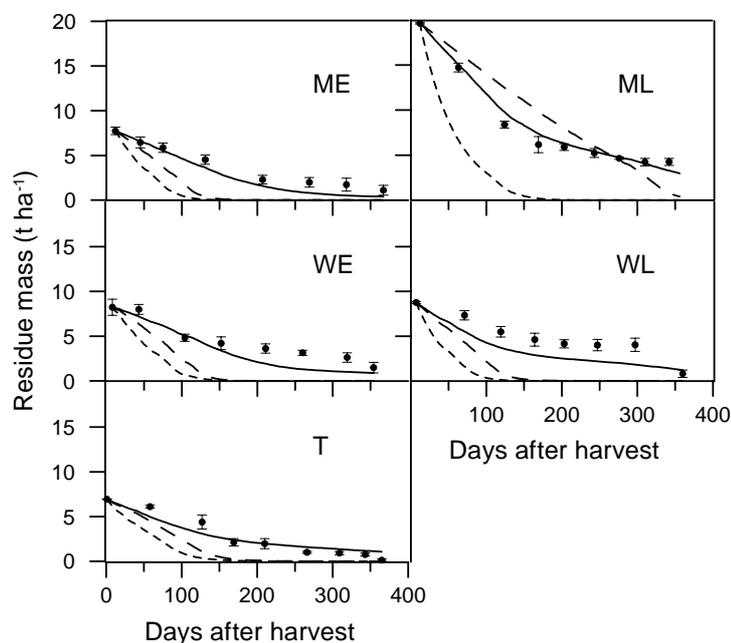
The parameter values were constrained during the optimisation to $1 \leq R_{crit} \leq 10$, $0.001 \leq D_{max} \leq 0.1$ and $20 \leq T_{opt} \leq 35$. The upper constraint to R_{crit} was chosen to be greater than the largest masses of residue studied by Dimes (1996), Stott *et al.* (1990) and Stroo *et al.* (1989), but still within a range that could be tested by the experimental data. The upper constraint to T_{opt} was chosen as the maximum optimum temperature of the temperature functions compared by Schomberg *et al.* (1996). The lower constraint to D_{max} was chosen to be lower than the value used by Snow *et al.* (1999) in simulating decomposition of tree litter.

An objective function, O , was defined for each experiment as

$$O = \sum (|R_{p,i} - R_{o,i}| + 1)^2 / n \quad (8)$$

where $R_{p,i}$ and $R_{o,i}$ are the predicted and observed residue masses at the i th sampling time and n is the number of sampling times. This function is similar to the mean square error commonly used to assess accuracy of model predictions. However, 1 was added to the difference between predicted and observed residue masses to add weight to the differences when residue mass values $< 1 \text{ t ha}^{-1}$, which were common, and allow for unequal number of sampling times in the different experiments. Optimal values of the parameters were found by the generalised reduced gradient optimisation method of Fylstra *et al.* (1998) to minimise the sum of O for both the ME and ML experiments (i.e. $O_M = O_{ME} + O_{ML}$). These two experiments were chosen for the optimisation because of the contrasting initial mass of residue in the experiments (Figure 9) and the availability of soil moisture measurements for calculating E_{os} .

Figure 9. Measured (\bullet , \pm one standard deviation) and predicted (lines) residue decomposition for the five field experiments. The lines correspond to the default predictions (short dash), and predictions from optimisations 1 (long dash) and 4 (continuous) described in Table 4.



Initially, only the value of R_{crit} was optimised (referred to as optimisation 1). Then, an additional parameter, either T_{opt} or D_{max} (optimisations 2 and 3) was included in the optimisation to reduce the value of O_M . Finally values of R_{crit} , D_{max} , and T_{opt} were all optimised (optimisation 4).

To provide an indication of their generality, the optimal parameter values derived in each of the four optimisations were used to predict residue decomposition and calculate O at the other three experiments. The values of O for all five experiments were then summed (O_A). O_M and O_A were also calculated for the original model (i.e., using F_{contact}) and both the default parameter values and various values of R_{max} (equation 6). These latter calculations were undertaken to explore the sensitivity of decomposition to F_{contact} in comparison to $F_{\text{contact}, 2}$.

Results

Residue decomposition in all experiments followed an approximately exponential decline through time (Figure 9), with between approximately 2 % (experiment T) and 20 % (experiment ML) of the initial residue mass remaining at the final sampling time. These amounts of decomposition are in general agreement with the results of Spain and Hodgen (1994) at a site in the Herbert region. In the WL experiment, there was a large decrease in mass between the last two sampling times. This decrease had no obvious environmental explanation, such as the onset of wetter and/or warmer conditions. There also appeared to be an acceleration in decomposition between the last two sampling times in the ME, WE and T experiments. However, the effect was much smaller than in the WL experiment.

Using APSIM-Residue with the default parameter values and F_{contact} , complete decomposition of residue was predicted in all experiments by approximately 150 days after harvest (Figure 9). However, in all experiments, except Tully, significant quantities of residue (0.8-4.2 t ha⁻¹) were present on the soil surface at the final sampling, approximately 360 days after harvest. For the default predictions, the values of O_M and O_A were 55.3 and 110.8, respectively (Table 4).

Table 4. Parameter values and resultant values of the objective function for the two experiments at the Mackay site (O_M) and for all five sites (O_A) for four parameterisation schemes. Values of O_M and O_A for the default parameters (Probert *et al.*, 1998) are also shown.

Number	Optimised parameters	Optimal parameter values			O_M	O_A
		D_{max}	T_{opt}	R_{crit}		
	Default	0.100	20	---	55.3	110.8
1	R_{crit}	0.100	20	1.4	22.0	65.6
2	$R_{\text{crit}}, T_{\text{opt}}$	0.100	35	3.1	13.8	49.9
3	$R_{\text{crit}}, D_{\text{max}}$	0.017	20	10	9.3	28.3
4	$R_{\text{crit}}, T_{\text{opt}}, D_{\text{max}}$	0.025	30	10	5.9	20.1

In the first optimisation, when $F_{\text{contact}, 2}$ was used, the optimal value of R_{crit} was 1.4 t ha⁻¹, resulting in a function for $F_{\text{contact}, 2}$ similar to that shown in Figure 7. The value of O_M was reduced to 22.0 (optimisation 1, Table 4), considerably less than the value for the default model. However, this reduction in O_M was mainly caused by the improved prediction of decomposition in the ML experiment (Figure 9). In the other experiments, predicted decomposition was also slowed, but still resulted in considerably less residue after the second or third sampling time than measured. Nevertheless, this change in APSIM-Residue resulted in the value of O_A being reduced from 110.8 to 65.6 (optimisation 1, Table 4).

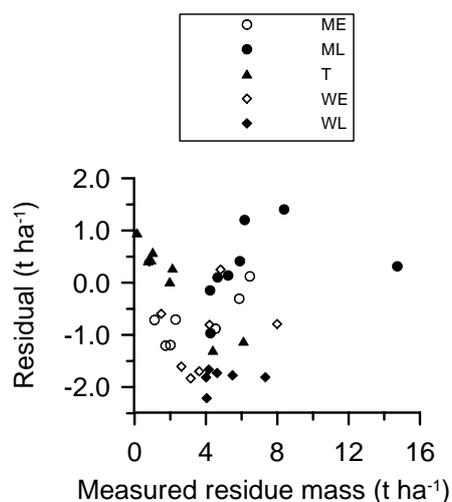
Inclusion of T_{opt} in the second optimisation improved predicted decomposition (data not shown). The optimal value of T_{opt} was 35°C (compared with a default value of 20°C), which was the maximum value chosen for the constraints of T_{opt} . With the inclusion of T_{opt} , the optimal value of R_{crit} , increased from that found in optimisation 1 to 3.1 t ha⁻¹. Compared

with optimisation 1, the values of O_M and O_A were considerably decreased (optimisation 2, Table 4).

Optimisation of the values of both R_{crit} and D_{max} resulted in substantially improved predictions of decomposition (data not shown). The value of O_M was 9.3 (optimisation 3, Table 4), less than half that in optimisation 1. There was a similar reduction in O_A . The optimal value R_{crit} (10 ha^{-1}) was higher than in either of the previous optimisations and at the limit chosen for its constraints. The value of D_{max} (0.017) was substantially lower than the default value (0.1).

Optimisation of the three parameters, R_{crit} , D_{max} and T_{opt} , further improved predictions of decomposition (Figure 9), with the improvements in O_M and O_A being approximately 30 and 40 %, respectively, compared with optimisation 3 (optimisation 4, Table 4). Unlike the previous two optimisations, the optimal values of T_{opt} was within the limits set for the constraints and the value of D_{max} was similar to that used by Snow *et al.* (1999). Visually, the poorest agreements between predicted and measured decomposition occurred between approximately 200 and 300 days after harvest in the two experiments at the Maclean site (WE and WL). This period occurred in different seasons in these two experiments because of the different harvest dates (July for WE and November for WL). So the discrepancy can not be simply attributed to the cooler climate at this site (Table 2). For this optimisation, there were no marked patterns in the residuals (Figure 10) but there was an average under-prediction of 0.53 t ha^{-1} . This under-prediction was largely due to the differences between measured and predicted decomposition at the Maclean site, described above.

Figure 10. Values of residuals as a function of measured residue mass for predictions made with the parameters derived in optimisation 4.



Discussion

Sugarcane trash deposited on the soil surface of the study sites at harvest mostly, but not completely, decomposed during the following year. The amount of decomposition was generally greater than that found in a study in the Herbert regions by Spain and Hodgen (1994). The amount of trash present a year after deposition was not a simple function of climate, as similar or greater amounts of trash were present at warmer, wetter sites (such as ML and T, Figure 9) as at the cooler, drier sites in NSW (WL and WE, Figure 9). It might be expected that retention of crop residues would promote a fungal-dominated, faster decomposition process than occurs in bacterial-dominated decomposition in tilled systems, as fungi numbers are very sensitive to residue inputs (Wardle and Lavelle, 1997). So the time for which the different sites had been trash blanketed may have impacted on the rate and amount of trash decomposition. There may be some evidence of this process occurring at the

NSW (WL and WE) sites, which had been trash blanketed for no more than one season prior to the experiment. At these sites there were relatively lower amounts of decomposition (i.e. greater mass of trash remaining) measured at 200-300 days after harvest (Figure 9). However, decomposition at ~ 350 days after harvest at these two sites was similar to or greater than that at the other sites, or reported by Spain and Hodgen (1994), suggesting that trash decomposition across these sites was not a simple function of the time for which the sites had been trash blanketed.

There are many other factors that can impact on the decomposition of surface residues. Biocide chemicals can reduce the number of macro-invertebrates in sugarcane farming systems (Robertson and Bakker, 2004) which, in turn, could reduce the burial of surface residues. The type and intensity of tillage that occurs, e.g. for weed control in a trash burnt system or at the end/start of a crop cycle, will also influence soil biology (Thompson, 1992; Tian *et al.*, 1997), as does fallow management (Pankhurst *et al.*, 2001). Information on these factors was not recorded at any of the sites studied in this experiment, so their impact can not be assessed.

Despite the lack of information on specific soil biological processes, decomposition was able to be described reasonable well (i.e. an average error of $< 2 \text{ t ha}^{-1}$) across all the sites with the modifications made to APSIM-Residue. Predictions of the decomposition of sugarcane residue were improved by the adoption of a new contact factor (equation 7) in APSIM-Residue. The improvement was particularly evident in the experiment (ML) where the initial mass of residue was large, $\sim 20 \text{ t ha}^{-1}$ (Figure 9). The residue blanket is $\sim 0.5 \text{ m}$ thick at this mass, so it is reasonable to assume that the upper trash layer will undergo negligible decomposition. Similar amounts of residue are common following harvest of large unburnt-sugarcane crops (Wood, 1991) so this improvement, alone, substantially enhances the capability of APSIM-Residue for application in sugarcane systems. The new contact factor has the advantage of fewer parameters (R_{crit}) than there were in the original contact factor (R_{min} , R_{max} and $F_{\text{contact, min}}$). However, this change alone did not improve predictions when residue mass was less than $5\text{-}7 \text{ t ha}^{-1}$ (Figure 9). At this mass (i.e. after approximately 100 days after harvest in all experiments except ML), predicted residue mass was still much lower than that measured in the experiments. New values of the parameters T_{opt} and D_{max} were also needed to give accurate predictions of decomposition in all experiments (Table 2).

There are two possible impacts of overestimating decomposition of large masses of residue when simulating sugarcane systems. The first is the simulation of immobilisation of soil N as the high C:N ratio residue decomposes too rapidly during simulations. Accurate simulation of residue decomposition, particularly of masses $> 5 \text{ t ha}^{-1}$, will avoid over-prediction of immobilisation. The second is the effect of surface residue on simulations of evaporation and runoff from the soil, through the relationship between residue mass and ground cover (Probert *et al.*, 1998). Sugarcane residue provides more than 90 % ground cover at masses greater than approximately 3 t ha^{-1} . However, ground cover is less than 20 % at masses below 0.3 t ha^{-1} , the mass that was predicted by the optimised 1 parameter values in all experiments (except ML) approximately 100 days after harvest (Figure 9). Thus predicted cover, and hence evaporation and runoff from the soil, will be markedly improved by the adoption of the new contact factor and sugarcane-specific values of T_{opt} and D_{max} . The effect of these modifications can be improved prediction of sugarcane yields.

The values of D_{max} needed to be reduced to more accurately simulate decomposition in the four experiments with lower initial masses of residue. When the value of T_{opt} was increased from the default to 30°C , the optimal value of D_{max} was similar to that used by Snow *et al.* (1999) to simulate the decomposition of tree (*Eucalyptus grandis*) leaves in an effluent

irrigated plantation. It would be expected that tree leaves would decompose more slowly than cereal or legume crop residues of a similar C:N ratio because of their generally higher contents of recalcitrant C compounds (Heal *et al.*, 1997). Thus, the need for Snow *et al.* (1999) to slow decomposition predicted by APSIM-Residue through the reduction of D_{\max} is understandable. The similarly low optimal value of D_{\max} found for sugarcane implies that there is some inherent factor limiting its decomposition that is not described by $F_{C:N}$, which accounts for residue “quality”. Marschner and Noble (2000) found that sugarcane residues, that had C:N ratios (96) similar to those in the five experiments (Table 2), decomposed slower than the leaves of two tree species, *Melia azedarach* and *Castanea sativa*, with C:N ratios of 44 and 103, respectively. These results support the adoption of lower D_{\max} values for predicting the decomposition of sugarcane residue. Another possible explanation for the slow decomposition of sugarcane residue in the field experiments is the lack of available soil N to satisfy the immobilisation demand of the high C:N ratio residue. However, there are several reasons why it is unlikely that this was the reason for the slow decomposition. In all experiments, the rate of N fertiliser application was likely to have been at least 50 kg ha⁻¹ in excess of the crops’ requirements. Secondly, decomposition of residue with the highest C:N ratio, 117 at Tully (Table 2), was not markedly slower than in the experiment with the lowest C:N ratio (76 in experiment ME). Thirdly, addition of N to sugarcane residue in laboratory studies has failed to significantly speed decomposition (F.A. Robertson, unpublished data). Still, the possible N limitation to decomposition of sugarcane residue should be considered more explicitly.

8. THE IMPACT OF TRASH BLANKETING ON DENITRIFICATION

Introduction

Denitrification has important repercussions, both beneficial and detrimental, for agricultural production and environmental quality. In current agricultural systems fertiliser sources are frequently used inefficiently, typically resulting in less than 50% of the applied N being up taken by plants (Mosier *et al.*, 2002). So, the removal of excess nitrate from the soil by denitrifiers can provide aid in the protection of ground and surface water through the reduction of nitrate leaching (Barton *et al.*, 1999). However, apart from economic losses, gasses emitted during denitrification are greenhouse gasses, so this process should be minimised wherever possible. Denitrification has been shown to be a potentially important pathway for loss of N from a range of soil types in sugarcane cropping systems (Weier *et al.*, 1996).

Numerous soil conditions have been cited in the literature as major controllers of denitrification rates. Denitrifying microorganisms are widely distributed, thus are not generally considered a limiting factor for denitrification (Barton *et al.*, 1999). Instead soil O₂, NO₃ and C concentrations, which have a direct effect on the activity of the microbes, are considered key controllers of denitrification (Barton *et al.*, 1999). The management of crop residues will have an affect on these controllers. Where residues are retained, soils are likely to be wetter and possibly have higher C concentrations. Avalakki *et al.* (1995a, b) measured higher denitrification from a vertisol with wheat and sorghum residues retained. Weier (1996) also measured increased N₂O emissions, one of the gasses released during denitrification, under trash blanketing. However, no data exist on the impact of trash management on denitrification from sugarcane cropping systems.

The processes controlling denitrification are transient, so it is difficult to extrapolate the finding from experiments, which are usually conducted over days or weeks, to longer time spans. Thus, considerable effort has been directed towards modelling denitrification from easily measurable soil parameters (Barton *et al.*, 1999). Typically this may include a combination of the following soil properties: soil moisture content (as gravimetric, volumetric or WFP), soil N, available soil C, soil temperature, pH, bulk density, field capacity, and CO₂ respiration (Barton *et al.*, 1999; Mosier *et al.*, 2002). Many models of soil N cycling include descriptions of denitrification. These can be coupled with crop growth models, giving extensive descriptions of N cycling in soil-plant systems. APSIM is an example of such a modelling system.

Cropping systems models such as APSIM need to accurately predict N losses from these systems. As noted above for APSIM-Residue, some APSIM modules have only been tested in grain and/or legume systems in semi-arid, rainfed areas (Carberry *et al.*, 1996; Probert *et al.*, 1995, 1998) and so could benefit from wider testing. This is true for the description of denitrification within the APSIM Soil-N module. The denitrification rate is a function of moisture, temperature, carbon and nitrate (NO₃) concentrations, based on the CERES-Maize V1 model. It has not been verified in situations where loss via denitrification could be significant, particularly in tropical regions.

This study aimed to (1) determine whether denitrification was increased by trash blanketing, and (2), using these data, improve the accuracy of denitrification modelling in the APSIM-SoilN module.

Methodology

Experimental

Site description

The field experimental site was situated on a sugarcane farm at North Eton (21° 14' S, 148° 58' E), 25 km west of Mackay. The area has a tropical climate with a mean annual rainfall of 1694 mm. Field measurements were conducted from the end of November 1996, following the crop harvest. Trash had been previously burnt at the farm so only short-term effects of trash on denitrification were explored in the study. The soil type was silty clay, classifying as an Ug 5.13 using the Northcote field classification system. Other soil characteristics are given in Table 5.

Table 5. Soil characteristics of the denitrification field experimental site.

Depth (cm)	pH ^a	Total N (%)	Total C (%)	Mineral – N (mg/kg)		Bulk Density (Mg m ⁻³)	DUL (%)	WP (%)	Particle size analysis (%)		
				NO ₃ -N	NH ₄ -N				Sand	Silt	Clay
0-5	5.7	0.15	1.61	3.7	3.3	1.22	39.8	21.1	27.7 ^b	24.5 _b	47.8 _b
5-10	5.5	0.12	1.63	0.9	3.0	1.36					
10-20	5.8	0.14	1.77	0.9	2.5	1.39	44.1	23.6	27.9	25.3	46.7
20-40	6.0	0.12	1.52	0.6	1.8	1.48	44.2	25.7	28.9	28.6	42.5

^a 1:5 soil/water

^b 0-10 cm

Field experiment

The field experiment consisted of 42 micro-plots, comprising a factorial of seven sampling times of two treatments (trash and no-trash), replicated three times. Micro-plots were placed within an area 14 cane rows wide (21 m) and 10 m long, which had been green cane harvested two days before the commencement of the study, leaving 10 t/ha of trash on the soil surface. Each row was halved and the trash removed from one half of each row to form the no-trash treatment. Each row contained six micro-plots (two treatments by three replicates). The micro-plots were PVC cylinders pushed into the soil to a depth of 20 cm, located in the row between cane stools. Potassium nitrate (KNO₃), at a rate of 160 kg N/ha, was broadcast on the soil surface of all micro-plots and watered into the soil, after which the whole experimental area was irrigated until waterlogged. Another irrigation event was conducted five days into the experiment to maintain soil moisture at contents conducive to denitrification. After the initial irrigation event, the mean water content in the micro-plots was 0.5 m³m⁻³, a value taken as the water content at saturation.

The procedure for obtaining and analysing samples from the micro plots followed that described by Weier (1999). The micro-plots were covered for one hour and gas samples taken from the headspace at the start and end of the hour for flux measurements. This procedure was repeated every eight hours for nine days. Gas samples were analysed for N₂O and CO₂

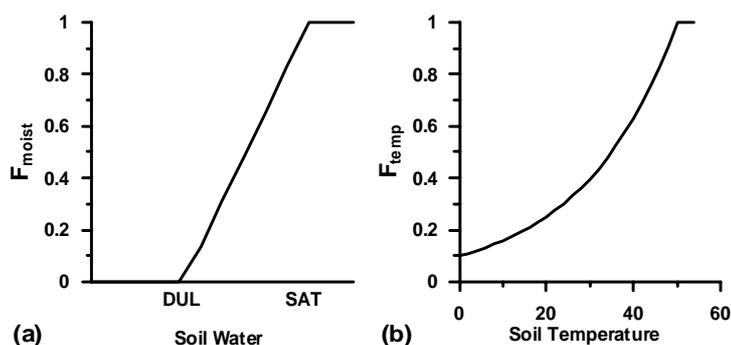
$$F_{moist} = \left(\frac{SW_{layer} - DUL}{SAT - DUL} \right)^1 \quad (11)$$

$$F_{temp} = 0.1 \exp(0.046 \times soil_temp_{layer}) \quad (12)$$

where, SW_{layer} is the volumetric water content for each layer, DUL and SAT is the volumetric water contents at the drained upper limit (or field capacity) and saturation respectively, and $soil_temp_{layer}$ is the temperature of the soil layer ($^{\circ}C$).

Figure 12.

The default functions for the two factors (equations. 11 and 12) that limit denitrification in APSIM-SoilN due to: (a) volumetric water content and (b) soil temperature.



A new moisture factor

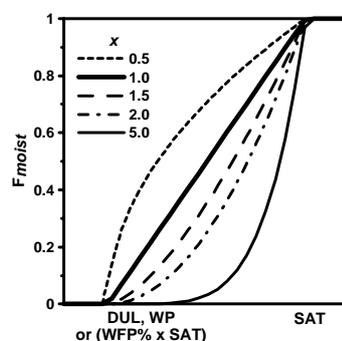
APSIM-SoilN module uses a linear function in the calculation of the F_{moist} (equation. 11 and Figure 12a) to provide a scale that runs from a threshold water content (DUL), below which no denitrification occurs, to an unconstrained rate at saturation (SAT). There are some weaknesses with this moisture factor. Firstly, experimental evidence suggests that a curvilinear response would be better suited (Weier *et al.*, 1993), and secondly the use of DUL does not recognise the issue of microsites that can cause denitrification to occur at water contents below the DUL. The following new moisture factor is proposed:

$$F_{moist} = \left(\frac{SW_{layer} - SW_1}{SAT - SW_1} \right)^x \quad (13)$$

where, SW_1 is the threshold water content below which there is no denitrification, SAT is the water content at saturation and x is a power between a lower limit of zero and an upper limit of five. In this study different values of SW_1 and x were investigated (Figure 13).

Figure 13.

The proposed new moisture factor (equation. 13) to limit denitrification in APSIM-SoilN due to volumetric water content.



Active, total and soluble carbon

The effect of carbon on denitrification (equation. 10) in the APSIM-SoilN module (version 2.1) is derived from the active carbon expression presented by Rolston *et al.* (1984).

$$“Active\ carbon = 24.5 + 0.0031 \times Soil\ Organic\ C” \quad (14)$$

This expression does not consider the susceptibility to decomposition of the different forms of carbon within the soil. The current expression used in the APSIM-SoilN module (equation. 10) replaces the “Soil Organic C” variable with two carbon fractions, HUM_C_{ppm} and FOM_C_{ppm}. There were concerns regarding this change, such as the impact of excluding the most labile carbon pool in the model (BIOM_C_{ppm}) and the justification for the constant, 24.5, which can be large when compared to the second term, meaning that significant denitrification can be predicted even when there is little carbon present.

Because of these issues, this study additionally investigated the use of two alternative definitions for the active carbon component used in the denitrification equation (equation. 9). As mentioned above, the current active carbon expression used in the APSIM-SoilN module (equation. 10) improved on Rolston *et al.*'s (1984) original expression by considering multiple C forms that are determined within the modelling environment. In an attempt to improve this component, this study investigated the replacement of HUM_C_{ppm} and FOM_C_{ppm} with total C concentration, a readily measured variable determined in the majority of experiments concerned with the soil environment. Essentially by using total C the third carbon pool used in APSIM-SoilN (BIOM_C_{ppm}) has been included. Thus making active C a function of total C:

$$C_A = 0.0031(HUM_C_{ppm} + FOM_C_{ppm} + BIOM_C_{ppm}) + 24.5 \quad (15)$$

$$C_A = 0.0031(totalC) + 24.5$$

Rolston *et al.* (1984) defined the active carbon calculated by equation 14 as “water extractable organic carbon”, which was interpreted to be equivalent to soluble carbon or dissolved organic carbon, *vis*:

$$C_A = SolubleC \quad (16)$$

Thus in this study we have also investigated the replacement of active carbon with measured soluble carbon acquired from the field experiment. This provided a means of removing the unknown consequences of the constants presently used in the calculation and made use of an easily measurable variable that could be built into a standard field data collection routine.

Model parameter optimisation

A numerical optimisation was undertaken to determine parameter values that would result in a more accurate prediction of denitrification in soils under sugar cane production. For the optimisation process, the denitrification equation utilised within the APSIM-SoilN module was converted into a simple spreadsheet model based on a daily time step, with the data required for the model being derived from field measurements.

Any measurements collected more frequently than daily were converted to a daily time step to correspond to the time-step used in the APSIM model. On days five and eight no measurements of soil NO₃, soluble C, total C and water content were made so their values were assumed to be the average of the measurements on the preceding and subsequent days. NO₃, soil water content, and total C were measured to 20 cm depth, while soluble C was only measured to 10 cm. Because of this, the depth to which the model was established was limited to 10 cm when soluble C was used in the determination of denitrification. This was not considered to be a significant limitation as other research has shown that denitrification below 10 cm depth is reduced by approximately an order of magnitude. This occurs because

of a lower number of facultative anaerobes in the subsurface soils (Luo *et al.*, 1998). The full depth of 20 cm was used where denitrification was determined using total C.

As discussed previously a new F_{moist} was considered in this study, containing a new variable SW_I to represent the water content below which no denitrification occurs. Four values for SW_I were tested, DUL, wilting point (WP), water content at 60% of saturation (0.60 x SAT) and the water content equivalent to an optimised percentage of saturation (WFP% x SAT).

Within the spreadsheet model, parameters of particular interest were isolated and focused on by the optimisation procedure. These included: the denitrification rate coefficient (k), the water factor power (x) used in the moisture factor equation (equation. 13), and WFP% used to define SW_I . Appropriate limits of the parameter values were specified prior to optimisation. Currently the APSIM-SoilN [version 2.1] code bounds x between a lower limit of zero and an upper limit of five, and these limits were used in the optimisation. WFP% used in the calculation of SW_I was set between the WFP% of field capacity (DUL) and wilting point (WP), 84% and 39% respectively, these being the highest and lowest WFP% for the soil depths considered. The “Solver” function in MS Excel 2000 was used to optimise the difference between the predicted denitrification results and that of the measured data. The Solver settings that were altered are listed in Table 6. The optimisation was performed on cumulative denitrification rather than the daily denitrification rate since longer-term N balances are generally of more interest than daily N flux in APSIM modelling. This was achieved by minimising the overall sum of the squared differences between the predicted and measured ($\sum E^2$) by changing one or more of the selected parameters.

Table 6. Solver settings altered in the optimisation.

Solver option	Setting adopted
Precision	0.000001
Tolerance	1%
Convergence	0.000001
Assume non-negative	Yes
Use Automatic Scaling	Yes
Estimates	Quadratic
Derivatives	Central
Search	Newton

As three forms of active carbon were investigated in this study each was given a unique identifier to distinguish them. These were “A” for active carbon as a function of FOM_C and HUM_C (equation. 10), “B” for active carbon as a function of total carbon (equation. 15) and “C” for active carbon replaced by soluble carbon (equation. 16). To represent a particular combination of parameter values, a number (e.g. 0-12) follows the letter (Table 7). Identifiers A0, B0 and C0 represent the standard APSIM parameter values for k , x , and $SW_I = \text{DUL}$ with only the active carbon definition changing for each letter. Scenario A0 represents the current

calculation used in APSIM-SoilN (version 2.1). Following this the denitrification rate coefficient (k) was optimised (referred to as optimisation 1). Then additional parameters, x and/or SW_I via WFP%, were included into the optimisation procedure to reduce the value of $\sum E^2$.

For the sake of simplicity root mean squared error (RMS) was used in the discussion of results instead of $\sum E^2$. As well as RMS, optimisations were assessed using the difference between the measured and predicted total cumulative denitrification over the whole experimental period (TCD) summed over both treatments. The optimisations can simulate the two treatments unequally well and therefore an absolute average of the TCD was used to summarise (i.e. if TCD-Trash = -150 and TCD-No trash = 300 then the absolute average TCD = 225). Another comparison was done on the ability of a scenario to match the cumulative treatment difference that was measured.

Table 7. The optimisation number, parameters optimised and changes in original water factor definitions in the 13 parameterisation schemes considered.

Optimisation number	k	x	SW_I
0	0.000600	1	DUL
1	Optimised	1	DUL
2	Optimised	Optimised	DUL
3	0.000600	1	WP
4	Optimised	1	WP
5	Optimised	Optimised	WP
6	Optimised	2	WP
7	0.000600	1	0.60 x SAT
8	Optimised	1	0.60 x SAT
9	Optimised	Optimised	0.60 x SAT
10	0.000600	1	WFP% ¹ x SAT
11	Optimised	1	WFP% ¹ x SAT
12	Optimised	Optimised	WFP% ¹ x SAT

¹ WFP% is an optimised value.

Results

Field data

Cumulative denitrification from the individual micro-plots from both treatments was highly variable (Figure 14). However, there was still significantly greater mean denitrification from the trash treatment, particularly from day two to day five (Figure 15). Over the whole experiment the average production of ¹⁵N-(N₂O+N₂) was 1045 and 851 g N/ha/day for the trash and no-trash treatments, respectively. This is equivalent to a total N loss through denitrification of 9.41 kg N/ha for the trash treatment and 7.66 kg N/ha for the no-trash treatment over the 8 days, a 1.63 kg N/ha (i.e. 20 %) treatment difference.

The greatest amount of microbial respiration, measured as CO₂ evolution, occurred in the trash treatment within the first day after irrigation (Figure 16). After this time the difference between the treatments was not significant. The second irrigation event after day five had no effect on the evolution of CO₂. There was no relationship found between denitrification and CO₂ evolution.

Figure 14.
Cumulative denitrification for individual cylinders grouped according to (a) Trash and (b) No-trash treatments. These are 8 hourly measurements.

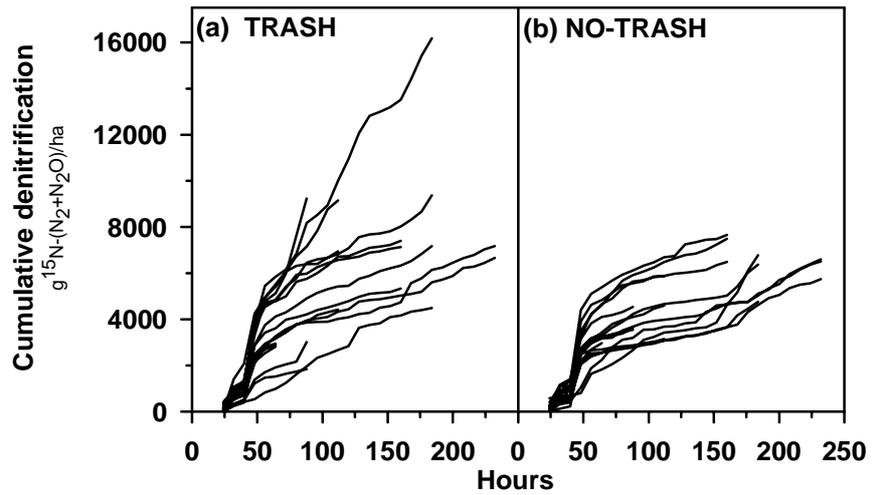


Figure 15.
The flux of ¹⁵N-(N₂+N₂O) average across treatments. The vertical bars show the standard errors of the mean. These are 8 hourly measurements.

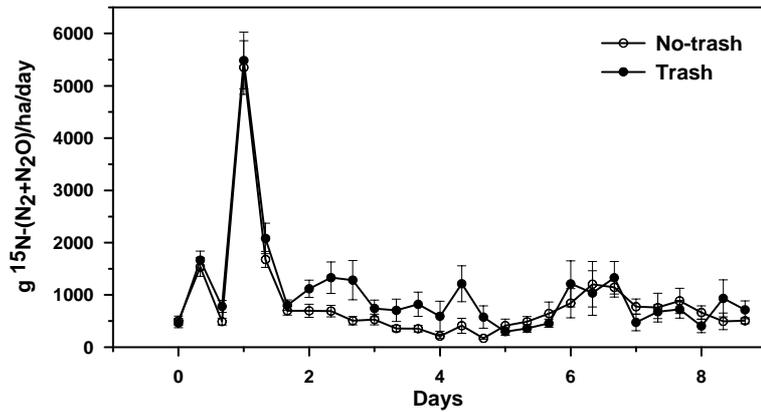
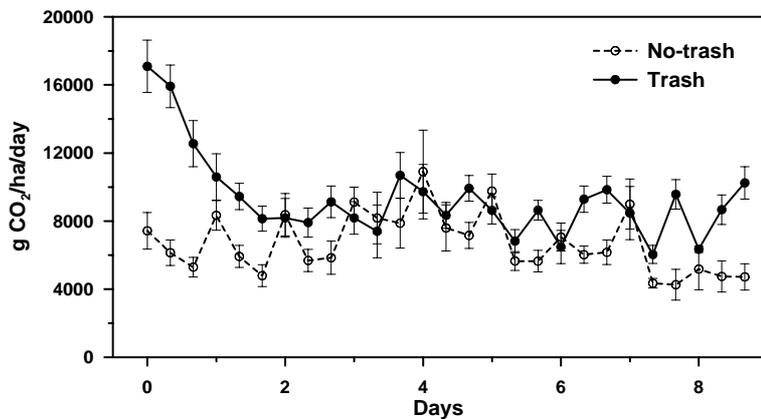


Figure 16.
CO₂ evolution averaged across treatments. The vertical bars show the standard errors of the mean. These are 8 hourly measurements.



Soluble C in the trash treatment was ~ 14 % greater than the no-trash treatment in the top 5 cm of soil (Figure 17). The average amount of soluble C in this layer under trash and no-trash treatments were 28.9 ± 1.7 ug C/g soil and 25.2 ± 1.7 ug C/g soil, respectively. In the next

layer of soil, 5-10 cm, the soluble C was similar for both treatments but declined over the period of investigation.

The soil water contents under trash were generally higher than the no-trash treatment, with the greatest difference occurring in the top soil layer (Figure 18). The addition of irrigation after five days increased the volumetric water content to 40% in both treatments for the top 5 cm of soil.

Soil temperature beneath the trash blanket was on average 1.4°C cooler (Figure 19) than the no-trash treatment. Over the experiment period the average temperature was 27.0°C and 28.4°C for trash and no-trash, respectively.

Figure 17. Soluble C for two soil layers measured destructively over the 8 days. The two layers were (a) 0-5 cm and (b) 5-10 cm.

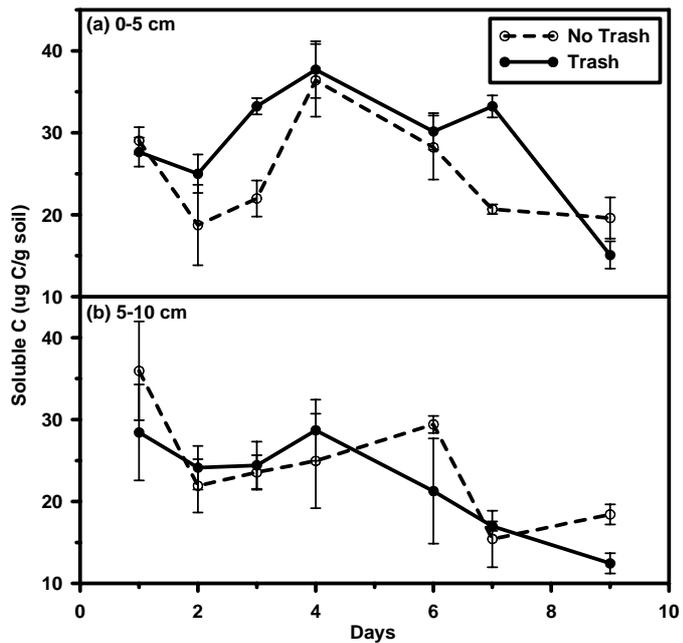


Figure 18. Volumetric water content for two soil layers measured destructively over the 8 days. The two layers were (a) 0-5 cm and (b) 5-10cm.

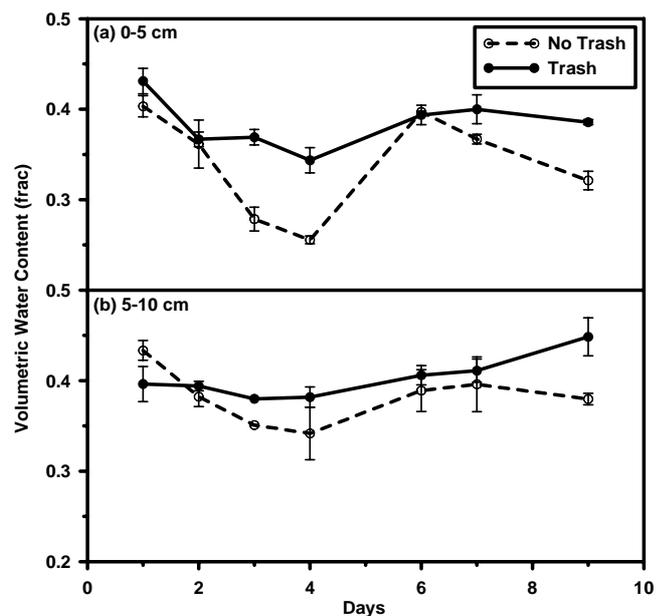
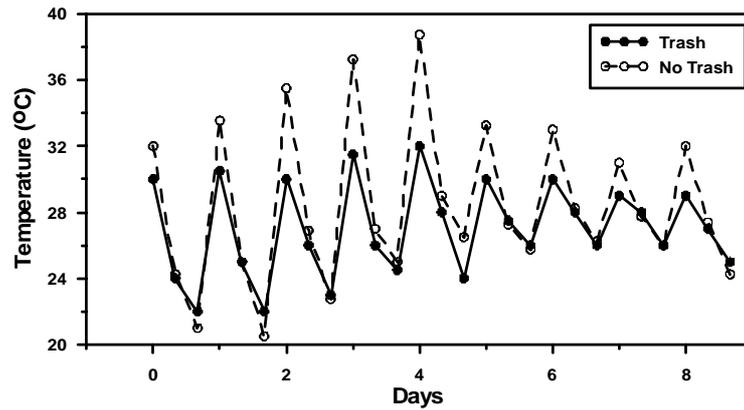
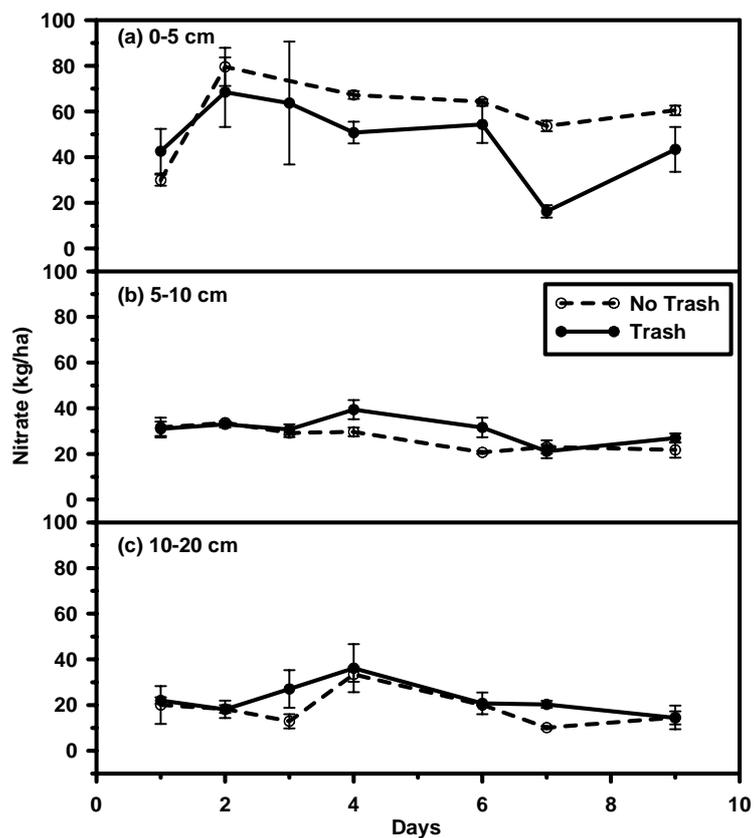


Figure 19.
Soil temperature measured every 8 hours over the 8 days.



Nitrate concentrations in the soil declined gradually over the period in each of the three layers (Figure 20) and the surface layer had approximately double the amount of nitrate compared to the lower depths. Interestingly, nitrate concentrations on the 0-5 cm layer were generally higher under no trash than the trash treatment, especially at the end of the experiment.

Figure 20.
Soil nitrate concentration over the 8 days for each layer.



Model optimisation for denitrification

Active C as a function of FOM_C AND HUM_C

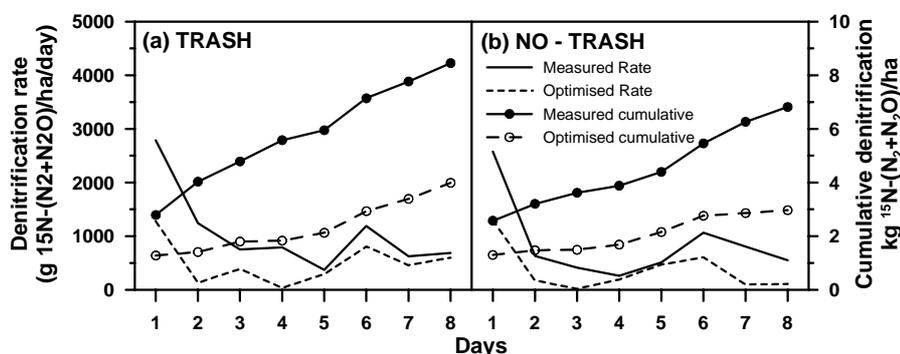
The resulting parameter values and errors for all parameterisation schemes using the default C_A definition (equation. 10) are presented in Table 8.

The current APSIM parameters (A0, Table 8) underestimated the denitrification under both treatments. In both treatments, the predicted total cumulative denitrification was equivalent to 50% of that measured. The rate of denitrification was underestimated on every day (Figure 21).

Table 8. Parameter values and resultant values of RMS for the APSIM default (A0) and 12 parameterisation schemes using C_A as a function of FOM_C + HUM_C. The bold values are the optimised parameters. The shaded cells highlight the smallest error terms.

Scenario #	Optimised Parameters	Parameter values (Optimised values are bold)			RMS	Total cumulative denitrification (TCD) (measured - predicted) (g N/ha)			Treatment Difference (kg N/ha) Measured = 1.63
		k (kg/mg)	x	SW_1		Trash	No-trash	Absolute average	
A0	none	0.000600	1.00	DUL	3124	-4461	-3849	4155	1.02
A1	k	0.001379	1.00	DUL	622	720	5	363	2.35
A2	k, x	0.001175	0.90	DUL	610	952	-136	544	2.72
A3	none	0.000600	1.00	WP	27169	51530	28541	40035	24.62
A4	k	0.000098	1.00	WP	1195	1383	-1020	1201	4.04
A5	k, x	0.000055	0.00	WP	1046	808	-255	532	2.70
A6	k,	0.000163	2.00	WP	1251	1612	-1300	1456	4.55
A7	none	0.000600	1.00	0.60 x SAT	15746	31955	14101	23028	19.49
A8	k	0.000151	1.00	0.60 x SAT	1310	1744	-1539	1641	4.92
A9	k, x	0.001718	4.98	0.60 x SAT	905	1339	-141	740	3.11
A10	SW1	0.000600	1.00	0.78 x SAT	1024	1825	-711	1268	4.17
A11	k, SW1	0.000637	1.00	0.78 x SAT	1019	1852	-569	1210	4.05
A12	k, x, SW1	0.001707	5.00	0.60 x SAT	905	1345	-149	747	4.05

Figure 21. Denitrification prediction using default APSIM parameters for each treatment (A0). Lines with symbols represent cumulative denitrification.

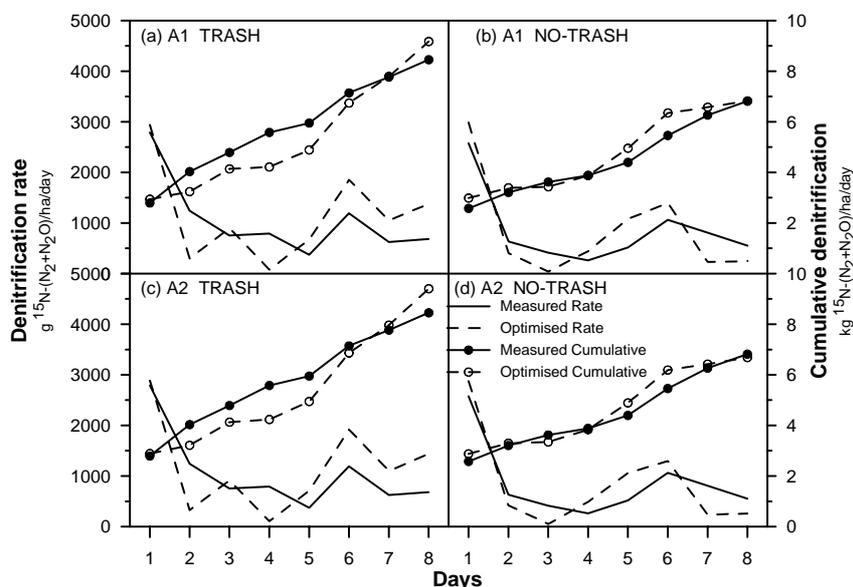


With SW_1 set to DUL, optimisation of k greatly improved the prediction of denitrification with a RMS of 622 (A1, Table 8) compared to a RMS of 3124 for the current APSIM parameters. In addition, A1 ($k = 0.001379$) produced the best prediction of the TCD for the scenarios using the current C_A definition (equation. 10). Optimising both k and x made a

small improvement in RMS but there was very little difference between optimisations A1 and A2 (Figure 22). The treatment difference in total denitrification was exaggerated by these scenarios (A1= 2.32 kg N/ha, A2 = 2.72 kg N/ha) but was the best of the new scenarios based on the current C_A definition.

No further improvement was achieved by changing SW_I to WP or 60% of SAT even with optimisation of both k and x (Table 8).

Figure 22.
Comparison of measured and optimised denitrification for a) A1 Trash, b) A1 No-trash, c) A2 Trash and d) A2 No-trash.



Active C as a function of total C

The resulting parameter values and errors for all parameterisation schemes using C_A based on total C are presented in Table 9.

Using total C to calculate C_A and the default values for k , x and SW_I (B0, Table 9), resulted in an under prediction of the TCD for both trash and no-trash treatments of 3946 g N_2+N_2O /ha and 1647 g N_2+N_2O /ha, respectively. This scenario did however improve on A0 discussed in the previous section.

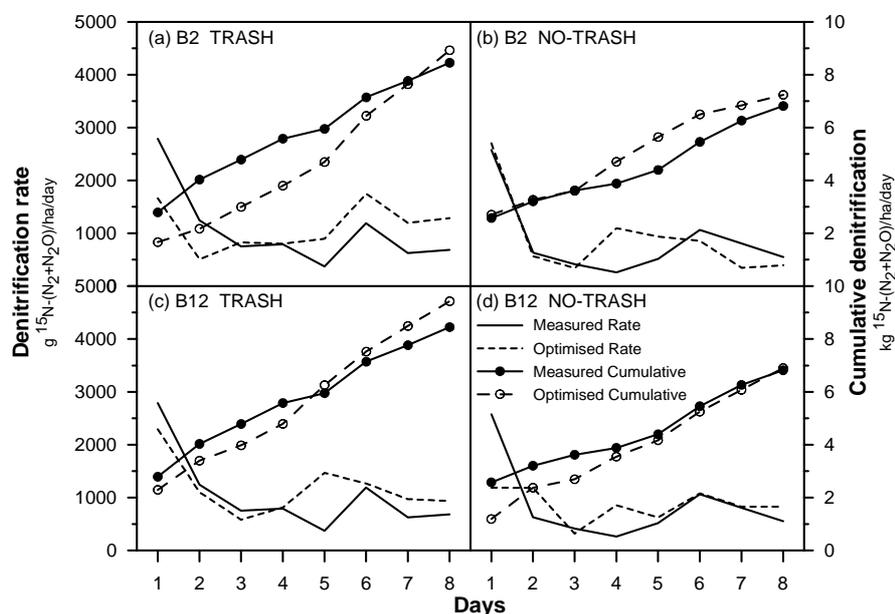
Scenario B12, where k , x and SW_I were optimised, produced the best RMS of 679. While B2 did not achieve a similar low RMS, the absolute average TCD was slightly better (448 g N/ha) and the treatment difference (1.69 kg N/ha) was very close to that measured. The similarity between these scenarios was obvious when the moisture factor, F_{moist} , was more closely investigated. For B12, F_{moist} performed as a switch, turning the rate of denitrification to a maximum at a water content equivalent to 78% of saturation or greater and below this water content, denitrification was stopped. This water content is comparable to DUL used in B2 and combined with a very low x (0.13) produced predictions much the same as B12 (Figure 23). Of the two scenarios, B2 predicted the final treatment difference the best.

Remaining scenarios tended to underestimate the initial peak in denitrification and then compensate by overestimating the denitrification on the following days. Thus resulting in a poor RMS but a good estimation of the total denitrification.

Table 9. Parameter values and resultant RMS for the APSIM default and 12 parameterisation schemes using C_A as a function of total C. The bold values are the optimised parameters. The shaded cells highlight the smallest error terms.

Scenario #	Optimised Parameters	Parameter values (Optimised values are bold)			RMS	Total cumulative denitrification (TCD) (measured - predicted) (g N/ha)			Treatment Difference (kg N/ha) Measured = 1.63
		k (kg/mg)	x	SW_1		Trash	No-trash	Absolute average	
B0	none	0.000600	1.00	DUL	2400	-3946	-1647	2797	-0.67
B1	k	0.000944	1.00	DUL	1446	-1368	1313	1340	-1.05
B2	k, x	0.000204	0.13	DUL	1038	475	420	448	1.69
B3	none	0.000600	1.00	WP	35658	56293	54493	55393	3.43
B4	k	0.000078	1.00	WP	1033	7	1193	600	0.45
B5	k, x	0.000262	3.51	WP	891	491	812	652	1.31
B6	$k,$	0.00132	2.00	WP	929	376	908	642	1.10
B7	none	0.000600	1.00	0.60 x SAT	20531	35180	29114	32147	7.70
B8	k	0.000125	1.00	0.60 x SAT	879	620	653	636	1.60
B9	k, x	0.000195	1.60	0.60 x SAT	868	633	647	640	1.62
B10	SW_1	0.000600	1.00	0.80 x SAT	1550	513	2153	1333	-0.01
B11	k, SW_1	0.000278	1.00	0.74 x SAT	765	765	369	567	2.03
B12	k, x, SW_1	0.000112	0.00	0.78 x SAT	679	978	85	531	2.53

Figure 23. Comparison of measured and optimised denitrification for a) B2 Trash, b) B2 No-trash, c) B12 Trash and d) B12 No-trash.



Active C replaced by Soluble C

The resulting parameter values and errors for all parameterisation schemes where C_A was replaced with soluble C are presented in Table 10.

Using measured soluble C and remaining with the default parameters for k , x , and SW_1 resulted in a significant under prediction of the denitrification rate (C0, Table 10). Understandably, almost all the other scenarios in this section improved on this.

The lowest RMS was achieved by scenario C12 while the best TCD was obtained by C5 (Table 10). Both scenarios had similar values for k and comparable F_{moist} curves achieved by using very different x and SW_1 values. Treatment differences were exaggerated for C5 (2.10 kg N/ha) and C12 (2.67 kg N/ha) and this was the case for most scenarios except C6, which simulated a treatment difference of 1.89 kg. In fact, scenarios C5-C12 all had similar RMS, TCD and treatment differences.

Careful application of these scenarios based on soluble C needs to be considered since the scenarios tended to underestimate the initial denitrification peak, particularly in the trash treatment, and then compensate by overestimating on the remaining days of the experiment (Figure 24). This may cause significant problems when extrapolated to a longer time period.

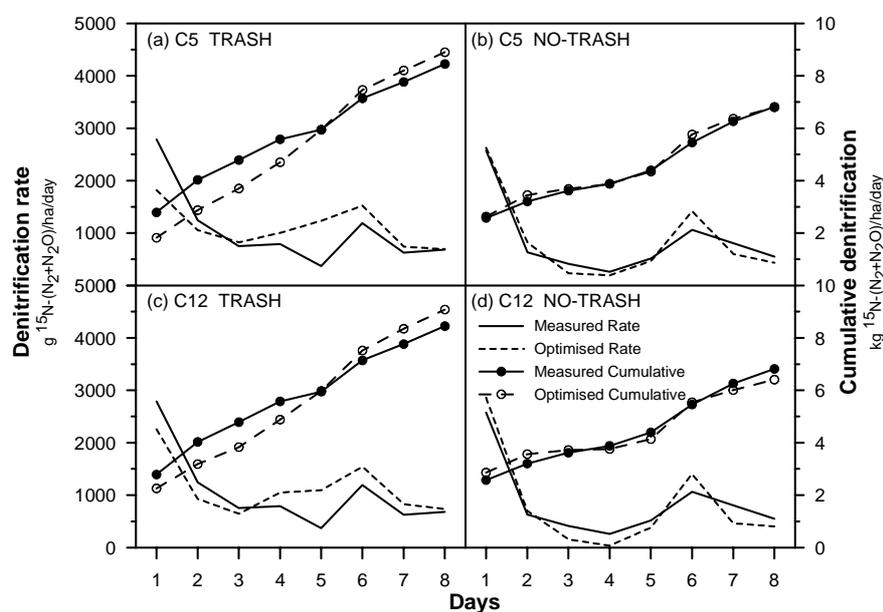
Table 10. Parameter values and resultant RMS for the APSIM default and 12 parameterisation schemes using soluble C instead of C_A . The bold values are the optimised parameters. The shaded cells highlight the smallest error terms.

Scenarios #	Optimised Parameters	Parameter values (Optimised values are bold)			RMS	Total Cumulative Difference (TCD) (measured - predicted) (g N/ha)			Treatment Difference (kg N/ha) Measured 1.63
		k (kg/mg)	x	SW_1		Trash	No- trash	Absolute Average	
C0	<i>none</i>	0.000600	1.00	DUL	4835	-7709	-6015	6862	-0.06
C1	k	0.004422	1.00	DUL	2137	-2996	-915	1955	-0.45
C2	k, x	0.001430	0.21	DUL	1615	-460	-1368	914	2.54
C3	<i>none</i>	0.000600	1.00	WP	5038	7943	7438	7691	2.14
C4	k	0.000312	1.00	WP	867	81	602	341	1.11
C5	k, x	0.001698	4.13	WP	553	448	-18	233	2.10
C6	$k,$	0.000571	2.00	WP	718	455	199	327	1.89
C7	<i>none</i>	0.000600	1.00	0.60 x SAT	783	1346	354	850	2.63
C8	k	0.000560	1.00	0.60 x SAT	681	693	-124	408	2.45
C9	k, x	0.001398	2.08	0.60 x SAT	530	522	-146	334	2.30
C10	SW_1	0.000600	1.00	0.61 x SAT	654	616	-199	407	2.45
C11	k, SW_1	0.001266	1.00	0.71 x SAT	491	677	-556	617	2.87
C12	k, x, SW_1	0.001317	1.16	0.70 x SAT	488	633	-407	520	2.67

Comparison of Active C definitions

Selecting one scenario from each of the three active carbon definitions the following achieved the best RMS values: A1, B2 and C5. Scenario A1 resulted in a marked improvement over the default parameters simply by changing the k to 0.001379. This provided an attractive option since no code change would be required to make these alterations. In comparison, B2 had a higher RMS value than A1, but at the same time maintained the treatment difference closer to that measured than A1 and C5. This was accomplished through the use of total C in place of HUM_C and FOM_C in the C_A formula, which can be achieved by including the third C pool (BIOM_C) defined within APSIM, and by changing k and x . Scenario C5 had the lowest RMS and best absolute average TCD of the these three scenarios, but had an exaggerated treatment difference. Only the top 10cm of soil was used in the soluble C optimisations (C0-C12) to achieve this result, compared to 20cm in the other scenarios. If soluble C is to be considered further than a comparison with the other C_A definitions should be done using an equal depth of soil for every scenario.

Figure 24.
Comparison of measured and optimised denitrification for a) C5 Trash, b) C5 No-trash, c) C12 Trash and d) C12 No-trash.



Discussion

The field experiment showed that trash blanketing has the potential to increase losses of N from sugarcane systems by denitrification, with N losses from the trash treatment 20 % greater than those with no trash. Overall, N losses during the experiment were equivalent to 6 % with trash and 5 % without trash of the N fertiliser applied at the start of the experiment. These results are in agreement with measurements of the impact of crop residue retention on denitrification in other cropping systems (Avalakki *et al.*, 1995a, b). The increases are also similar to the increased N₂O emissions caused by trash blanketing measured by Weier (1996). However, the site studied in this experiment did not have a history of trash blanketing. Long-term crop residue retention, including trash blanketing, impacts on soil biology (Tian *et al.*, 1997; Wood, 1991; Robertson *et al.*, 1995; Robertson and Bakker, 2004). Thus it is possible that denitrification rates measured in this experiment are not representative

from those areas with a long history of trash blanketing, and its associated impact on soil biology.

The increase in denitrification did not occur during the initial peak of denitrification that occurred following application of irrigation water and N fertiliser (Figure 15). Rather, it was following this peak when daily denitrification rates were close to their lowest during the experiment. The cause of the higher rate of denitrification under trash was not clear from the measurements of the soil and environmental properties controlling denitrification made during the experiment. Soluble C (Figure 17) and water contents (Figure 18) were higher in the trash treatment, but temperatures (Figure 19) and nitrate concentrations (Figure 20) were lower. These differences illustrate the complexity of the denitrification process, and the need for a mechanistic approach, in addition to an experimental approach, for determining the practical implications of the results for N fertiliser management in sugarcane systems.

The default method of modelling denitrification in the APSIM system underestimated denitrification from both treatments (Figure 21). Substantial improvements were achieved by increasing the value of the denitrification rate coefficient (Figure 22a and b). However, further, but only minor improvements could be achieved by adopting new values of other parameters, or through changing the definition of soil C in the denitrification algorithms.

9. IMPACT OF TRASH MANAGEMENT ON SOIL FERTILITY

Introduction

Crop residue retention is widely recognised as playing an important role in many agricultural production systems because of its diverse effects on soil physical, chemical and biological properties (Kumar and Goh, 2000). These changes to the soil properties may affect the soil environment in a way that results in an alteration of nutrient transformations and efficiency of use (Power and Doran, 1988). The degree to which these changes occur has been widely studied across various environments and in a plethora of cropping systems. The results of these studies generally demonstrate beneficial effects from crop residues retention, such as increased soil organic matter (SOM), enhanced soil structure, improvements in soil nutrients, and influences on soil moisture and temperature regimes (Kumar and Goh, 2000; Schomberg *et al.*, 1994; Lal, 1995). While many benefits of crop residue retention have been reported, the degree to which these occur and the time frame in which they are observed varies widely throughout the literature.

Despite the widespread adoption of GCTB in the Australian sugarcane industry over the past two decades, little is still known about the impact of GCTB on soil fertility. Trash blanketing has been shown to increase soil N and soil C (Wood, 1986, 1991; Blair *et al.*, 1998; Noble *et al.*, 2003), microbial biomass and associated N mineralisation (Sutton *et al.*, 1996) and labile C (Blair *et al.*, 1998; Noble *et al.*, 2003). These differences have been confined to shallow soil depths (1-2 cm in some cases; Blair *et al.*, 1998; Noble *et al.*, 2003). There has been little investigation of other nutrients: Exceptions being Wood (1986) who found no effect of trash blanketing on difference in soil K in the Herbert valley, and Noble *et al.* (2003) who found increased Ca, pH and acidity, but similar Mg, K and Na at a site in Tully. There has also been no attempt to coordinate investigations across different sites to establish the impacts of climate or soil type on the impacts of trash blanketing on soil fertility. Also, there has been no attempt to identify the time scale over which changes occur.

This study aimed to address these knowledge gaps by determining the magnitude and time scale of changes in soil fertility after the initiation of GCTB. This was achieved by measuring soil fertility parameters in five experiments comparing GCTB and burnt treatments. These experimental sites were located along the length of the sugar cane production region of the Australian east coast and had been under GCTB for up to 17 years, thus giving the study a broad spatial and temporal perspective.

Methodology

Sites

Soil samples were taken from five experimental sites located across the eastern span of the Australian sugar cane growing region, ranging from the North Queensland wet tropics through to northern New South Wales (NSW). Each site had been under GCTB management for varying lengths of time, ranging from 1 to 17 years, as well as differing in soil type, texture, site history, and average rainfall. Further site details are given in Table 11.

Three of the sites, Ayr, Mackay, and Tully, were replicated experiments located on BSES research stations. At the time of sampling both the Mackay and Tully sites were in their first crop cycle, while the Ayr site was in its second. The other two sites were located on

operational farms. The Woodford Island site was a replicated experiment established by Kingston *et al.* (1998) in the Harwood Mill area for BS168, and at the time of sampling had only one season (the first ratoon crop) of trash blanketing.

Wood (1986, 1991) established the Abergowrie site in the Herbert Valley as a GCTB demonstration site. At the time of sampling it was in its third crop cycle and, to our knowledge, was the oldest GCTB-burnt comparison site in Australia at the time of sampling. The experimental design was not randomised like the other four sites, but consisted of greater than 300 m long adjacent GCTB and burnt ‘strips’ of sugar cane, each strip being 10 rows (~ 16 m) wide. After harvest in 1997 the management of the burnt treatment changed due to the discontinuation of that treatment and the entire area converted to a GCTB management system. For the continuation of this experiment, trash was then raked by hand from four quadrats of approximately 25 m², at 10, 100, 200, and 300 m, from which all ‘burnt’ samples were taken for the remainder of the study. Due to the unreplicated design of the comparison, soil samples for the GCTB treatment were also taken at the same distances along the strip to give a four-paired comparison for statistical analysis.

Table 11. Details of sampled sites.

Site	Abergowrie	Ayr	Mackay	Tully	Woodford Is. (near Maclean)
Year planted	1980	1988	1992	1990	1995
Date sampled	10/98	11/98	8/98	11/97	1/98
Years of trash addition	17	9	5	6	1
Replicates	4	4	4	3	4
Treatment sampled	<ul style="list-style-type: none"> • GCTB • Burnt 	<ul style="list-style-type: none"> • GCTB • Burnt 	Early harvest: <ul style="list-style-type: none"> • GCTB-zero till • Burnt cultivated 	<ul style="list-style-type: none"> • GCTB-zero till • Burnt cultivated 	Early harvest: <ul style="list-style-type: none"> • GCTB • Burnt
Plot size	+600 m ²	+600 m ²	≈ 100 m ²	≈ 100 m ²	≈ 100 m ²
Soil texture†	Loam	Sandy loam	Loam/Clay Loam	Silty Clay	Silty Clay
ASC	Red Dermosol	Not available	Brown Chromosol	Brown Dermosol	Not available
Average rainfall (mm/yr)	1700	1080	1670	3480	1020
-33 kPa (%) *	18.94	14.33	21.24	30.83	33.93
-1.5 MPa (%) *	7.81	5.35	8.12	18.54	17.17

† Texture classes derived from PSA report. Textures were uniform between 5-10 cm and 25-30 cm depths, except for Mackay.

- 5-10 cm depth

Soil sampling

Soil samples were taken with 50 mm diameter steel ‘push tubes’ after the harvest period. For each plot a total of six cores were taken, three from the row and three from the inter-row, to a depth of 1.5 m. A further 12 cores to a depth of 25 cm were taken to provide additional soil for analyses done on the shallower depths. Samples from each core were divided into depth increments (0-2 cm, 2-5 cm, 5-10 cm, 10-25 cm, 25-50 cm, 50-75 cm, 75-100 cm, 1.0-1.25 m, and 1.25-1.5 m) and bulked accordingly. They were stored under refrigeration in plastic bags until analysis, which started within seven days of sampling.

Laboratory analysis

Carbon analysis

Total carbon (C_T) was determined to a depth of 1.5 m by dry combustion using a mass spectrometer. The fractionation of the total soil carbon was determined through the use of two different methods, the $KMnO_4$ oxidation technique and the UV photo-oxidation technique.

The $KMnO_4$ oxidation method was based on that described by Loginow *et al.* (1987), Lefroy *et al.* (1993), and Blair *et al.* (1995). In brief, the soil was reacted with an excess of $KMnO_4$ and the reduced permanganate, which is proportional to the amount of C oxidised, was measured by spectrophotometry (Blair *et al.*, 1998). Multiple $KMnO_4$ concentrations were utilised in this experiment due to the supposition that the lower the concentration required for oxidation the more labile the organic component.

The amount of oxidisable soil C was determined by two concentrations of $KMnO_4$ (33 mM and 333 mM $KMnO_4$), at 5 depths (0-2, 2-5, 5-10, 10-25, and 50-75 cm), for both treatments, and each replicate sampled. The final outcome was the representation of total C as four different fractions based on susceptibility to oxidation. Two C fractions were established through $KMnO_4$ oxidation – 1) the amount of C oxidisable by 333 mM $KMnO_4$, and 2) the amount oxidisable by 33 mM $KMnO_4$. Previous authors have identified these two fractions as ‘C3’ (Moody *et al.*, 1997;) or ‘labile C’ (C_L) (Conteh *et al.*, 1997, 1998, 1999; Blair *et al.*, 1995, 1998) and ‘C1’ (Moody *et al.*, 1997; Bell *et al.*, 1999), respectively. In an attempt to remain close to previously used nomenclature these two fractions are referred to in this paper as ‘ C_L ’ and ‘ C_{L1} ’, respectively (refer to Table 12).

Table 12. Summary of the nomenclature relationships used within the literature and that used in this paper.

References	333 mM $KMnO_4$	33 mM $KMnO_4$	Difference (333-33 mM) (i.e.167 mM)	Total C	Remaining C after oxidation ($C_T - C_L$)
Moody <i>et al.</i> , 1997; Bell <i>et al.</i> , 1999	C3	C1	C2	C_T	-
Conteh <i>et al.</i> , 1997, 1998, 1999; Blair <i>et al.</i> , 1995, 1998	C_L	-	-	C_T	C_{NL}
This paper	C_L	C_{L1}	C_{L2}	C_T	C_{NL}

The difference between C_L and C_{L1} produced another C fraction,

$$C_{L2} = C_L - C_{L1} \quad (17)$$

represents the amount of C oxidised by the additional KMnO_4 available at the higher concentration and was referred as ' C_{L2} ' (refer to Table 12). This essentially splits the C_L fraction into two parts, with C_{L1} representing the most readily labile C and C_{L2} the remaining labile C.

A final fraction was derived from the difference between total C and labile C,

$$C_{NL} = C_T - C_L \quad (18)$$

reflecting the amount of soil C that remained unoxidised by the KMnO_4 . Other researchers have referred to this fraction as the amount of non-labile C present in the soil, identifying it as ' C_{NL} ' (Conteh *et al.*, 1997, 1998; and Blair *et al.*, 1995, 1998), which was continued in this paper (refer to Table 12). The division of C into these fractions enabled the total soil C content to be considered as the sum of three fractions, C_{L1} , C_{L2} , and C_{NL} ,

$$C_T = C_{L1} + C_{L2} + C_{NL} \quad (19)$$

each reflecting the tendency of the contained carbon to oxidise. Refer to Table 13 for a summary of the calculated C fractions.

Table 13. Details of the C fractionation attributes analysed.

Method	Attribute	Description	Derivation	Replicates	Lower Depth
KMnO ₄ oxidation technique	C_T	Total soil C content	Dry combustion	Individual reps analysed	1.5 m
	C_{L1}	Amount of C oxidised by 33 mM KMnO_4	Oxidation with 33 mM KMnO_4		75 cm
	C_L	Amount of C oxidised by 333 mM KMnO_4	Oxidation with 333 mM KMnO_4		75 cm
	C_{L2}	The additional C oxidised when the KMnO_4 concentration increased from 33 to 333 mM (i.e. that oxidised by 167 mM)	$= C_L - C_{L1}$		75 cm
	C_{NL}	Non-labile C content; the unoxidised soil C component	$= C_T - C_{L2} - C_{L1}$ (i.e. $C_T - C_L$)		75 cm
UV photo-oxidation technique	POC	Particulate organic carbon; C content of OM greater than 53- μm	$= C_T - <53\text{-}\mu\text{m}$ fraction	Reps bulked and resultant sub-sample analysed	25 cm
	<53- μm fraction	C content of OM less than 53- μm	Oxidation of OM <53- μm		25 cm
	Inert C	Amount of C present after UV photo-oxidation of the <53- μm fraction; a sub-fraction of <53- μm fraction	Oxidation of OM <53- μm using UV photo-oxidation		25 cm
	Humic C	The UV oxidisable C component of the <53- μm fraction; a sub-fraction of <53- μm fraction	$= <53\text{-}\mu\text{m}$ fraction - Inert C		25 cm

Based on the C_L and C_{NL} fractions an additional index was calculated as a means of estimating the turnover rate of C in the system (Blair *et al.*, 1995). This is referred to as the '**lability of C**' (**L**), and expresses the amount of labile C (C_L) relative to the amount of non-labile C (C_{NL}) present in a given soil (Conteh *et al.*, 1997) and is given by,

$$L = C_L / C_{NL} \quad (20)$$

This index has a role in the determination of the continuity of C supply, along with the size of the C pool (Blair *et al.*, 1995).

The second C fractionation method, the UV photo-oxidation technique, was based on that described by Skjemstad *et al.* (1993, 1999). Again this resulted in multiple C fractions, which were calculated for both treatments, at four depths (0-2, 2-5, 5-10, 10-25 cm), on soil samples that were bulked across replicates and sub-sampled. The samples were first physically fractionated using a 53- μm sieve into two fractions. The first, the '**<53- μm fraction**' (labelled as such after Skjemstad *et al.*, 1999, 2001), represented the C content of the material that passed through the 53- μm sieve while in suspension. The second, the >53- μm fraction, reflected the C content of the organic material retained on the sieve, and was calculated as the difference between C_T and the <53- μm fraction.

$$POC = C_T - < 53\mu M \quad (21)$$

This fraction has been shown to consist largely of particulate organic matter (POM) and sand (Skjemstad *et al.*, 1999) and was defined as **particulate organic carbon** ('**POC**') (after Skjemstad *et al.*, 2001) to reflect this nature.

The material that was <53- μm was further fractionated using high energy UV photo-oxidation, resulting in two more C fractions. The C that remained after this further oxidation represented the C resistant to UV-oxidation in the soil, and was labelled '**inert C**' (after Baldock and Skjemstad, 1999).

The final fraction was calculated as the difference between the <53- μm fraction prior to photo-oxidation and the resultant inert C fraction, and was labelled '**humic C**' (after Skjemstad *et al.*, 2001).

$$Humic\ C = < 53\mu M - Inert\ C \quad (22)$$

This fraction represented the UV-oxidisable component of the total soil C that was <53- μm . Again the final result of fractionation was that the total carbon content could be considered in three components, POC, humic C, and inert C,

$$C_T = POC + Humic\ C + Inert\ C \quad (23)$$

on which treatment differences were observed. Refer to Table 13 for a summary of the calculated C fractions.

Nitrogen analysis

Total nitrogen (N_T) was determined to a depth of 1.5 m by dry combustion with a mass spectrometer. Potential N mineralisation for only the Abergowrie and Ayr sites was determined using four different methods to a depth of 25 cm (0-2, 2-5, 5-10, 10-25 cm). Aerobic and anaerobic mineralisation was established by measuring the amount of $\text{NO}_3\text{-N}$ and $\text{NH}_4\text{-N}$ mineralised during seven-day incubations under moist and constant temperature conditions, in near field capacity and water logged circumstances, respectively (after Bremner, 1965). Microbial biomass was measured by placing 15 g of soil in an evacuated desiccation with chloroform for 24 hours, after which the chloroform was removed and the

sample underwent mineral N extraction. The final method was hot mineral N extraction (Gianello and Bremner, 1986), using 3 g of soil in 20 ml of 2M KCl, heated in a block digester at 100°C for 4 hours, cooled, filtered and measured. NO₃-N and NH₄-N in the soil cores (to 1.5 m depth) and the four methods described above were determined colorimetrically (Henzell *et al.*, 1968) after extraction from the soil with 2M KCl (Catchpoole and Weier, 1980).

Other laboratory analysis

Exchangeable cations (Ca, Mg, Na and K) were determined using the 0.02M HCl BSES cation method as described by Haysom (1982). Other soil chemical properties were analysed by using the procedures described by Rayment and Higginson (1992). This included: pH and electrical conductivity (EC) in a 1:5 soil/water suspension; calcium phosphate-extractable sulfur (S); acid-extractable phosphorus (P); and cation exchange capacity (CEC) by 0.01M silver-thiourea (AgTU)⁺. All analyses were conducted on the depth increments of 0-2, 2-5, 5-10, 10-25 cm, for both treatments and all replicates.

Two soil physical aspects were also considered, these being the soil particle size and moisture retention characteristics. A particle size analysis (PSA) was conducted using the hydrometer-reciprocating shaker method described by Thorburn and Shaw (1987). The moisture characteristics of the soils were determined through the use of the standard gas pressure plate method, estimating the field capacity (-33 kPa) and wilting point (-1.5 MPa) of the soil samples. Both analyses were performed on sub-samples taken from the depth increments of 5-10, 25-50, 75-100, 125-150 cm and bulked across replicates.

Statistical analysis

Statistical analysis sought to address the question of whether the different management practices of GCTB and burning had an effect on soil chemical properties. As samples were taken from sites with varying time lengths of GCTB and for the full range of depths to 1.5 m, it was possible to also consider if there were trends associated with increased time under a management practice or with depth.

The soil chemical data for the Ayr, Mackay, Tully and Woodford Is. sites was analysed by a split-plot analysis of variance and the LSD method for comparison of means ($p < 0.05$) with a 'treatment x depth' interaction. Due to the unreplicated nature of the Abergowrie site, the data from this site was analysed using a paired T-test, with $P < 0.05$.

Because the bulking of soil replicates prior to examination of C using the UV photo-oxidation method, a statistical analysis on a site-by-site basis was unable to be performed for this C fraction technique. However, the data were used to observe visual trends and were included in a Pearson correlation analysis. A correlation analysis was conducted on the replicate means of the C fractions from both fractionation techniques to assess the degree of linear association between the variables. Scatter plots were used prior to correlation analysis to observe data trends and identify any anomalies.

All statistical analyses were determined using the analytical software package Statistix 7.

Results

Soil physical analyses

The results from the soil particle size analysis (Table 11) were used to provide a formal classification of the soil texture in the upper soil surface of each site. Also outlined in Table 11 are the soil moisture retention characteristics, field capacity (-33kPa) and wilting point (-1.5Mpa), of each site for the 5-10 cm depth. Differential measurements of soil physical properties were not affected for trash management treatments.

Carbon analysis

Total carbon (C_T)

There is a general trend for higher C_T contents in the surface soil layer for the GCTB treatment at all sites. Compared to the burnt cane treatment, the GCTB treatment was found to contain significantly higher amounts of C_T at 0-2 cm depth for the Ayr, Mackay, and Tully sites ($p < 0.05$) (Figure 25a). This trend was observed at the Abergowrie site, however it was only found to be significant at a higher level of significance ($p < 0.1$). One possible explanation for this result may be the different trial design and hence the different sampling procedure, which may not fully account for the spatial variability of the soil. The difference between treatments at these four sites (0-2 cm) were 0.11%, 0.21%, 0.14%, and 0.67% for Abergowrie, Ayr, Mackay, and Tully respectively.

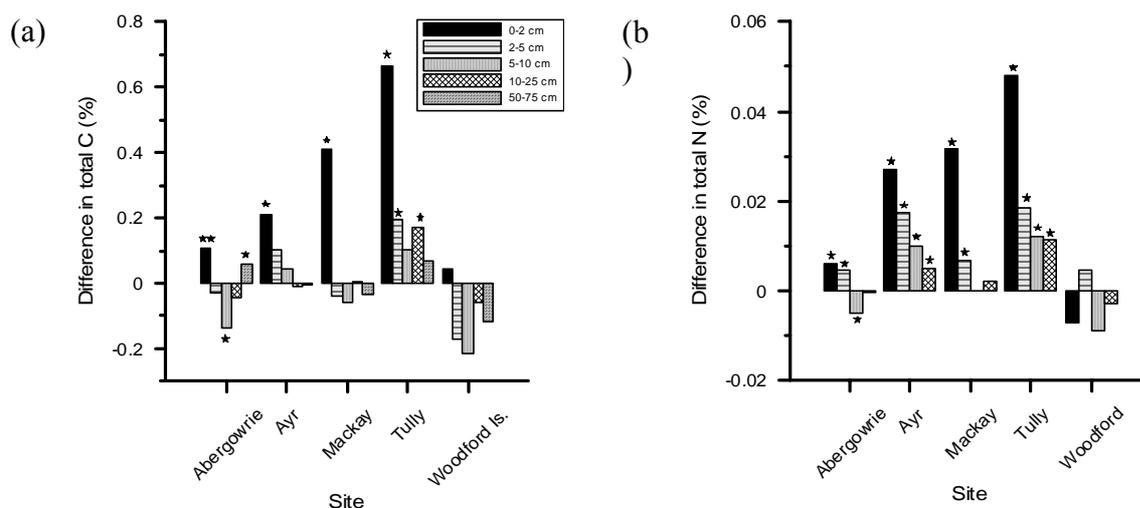


Figure 25. The differences (GCTB-burnt) in (a) total C and (b) total N for the five study sites. * Indicates a significant difference at $p < 0.05$. ** Indicates a significant difference at $p < 0.1$.

The Woodford Is. site showed no significant difference between treatments at any depth but did follow the trend for higher C_T contents in the surface soil (0-2 cm) of the GCTB treatment (an increase of 0.04%). The lack of difference at the Woodford Is. site may be attributed to its recent change to GCTB, with the site having experienced only one season of green cane trash management prior to soil sampling. This result agrees with that found by Ball-Coelho *et al.* (1993) who reported no measurable difference in total C between trash retained and post-harvest burnt treatments after just one year of comparison.

For other depths sampled, only the Abergowrie and Tully sites returned significant differences between treatments. The GCTB treatment had significantly more C_T at Tully in the 2-5 cm, and 10-25 cm depths ($p < 0.05$; differences of 0.19%, and 0.17%, respectively)

and Abergowrie at 50-75 cm ($p < 0.05$; difference of 0.06%). Abergowrie also returned the only significantly higher C_T result for the burnt treatment ($p < 0.05$), occurring at the depth of 5-10 cm (difference of 0.14%) (Figure 25a).

Generally the results indicate that trash retention causes an increase in the total soil C content, but essentially this increase is confined to the upper 2 to 5 cm of soil. The results presented no obvious trend for a proportional increase in C_T content of the surface soil (0-2 cm) with increased time of trash retention. The greatest increase in C_T occurred at Tully followed by Mackay, Ayr, Abergowrie, and Woodford Is., with the years under GCTB management reflected as 6, 5, 9, 17 years, and 1 year, respectively.

KMnO₄ oxidation technique

There was a general trend for the trash retention treatment to contain a greater concentration of the labile C fraction (C_L) compared to the burnt cane treatment, especially in the upper soil layers (Figure 26a). This trend was observed at all sites, except for Woodford Is., with the GCTB treatment containing significantly ($p < 0.05$) more C_L at Abergowrie (to 5 cm and in the 10-25 cm depth), Ayr (to 10 cm) and Mackay (to 2 cm) (Figure 26a). Tully followed this trend but no significant differences between the two treatments were found. In contrast, a significantly ($p < 0.05$) greater concentration of C_L was observed in the burnt cane treatment at the Woodford Is. site (0-2 cm) compared to the GCTB treatment. The trend for higher C_L in the GCTB treatment indicates that, as would be expected, trash retention has a tendency to input more labile C into the system than cane burning. The exception observed at the Woodford Is. site may be explained again by its recent change over to a trash retention system.

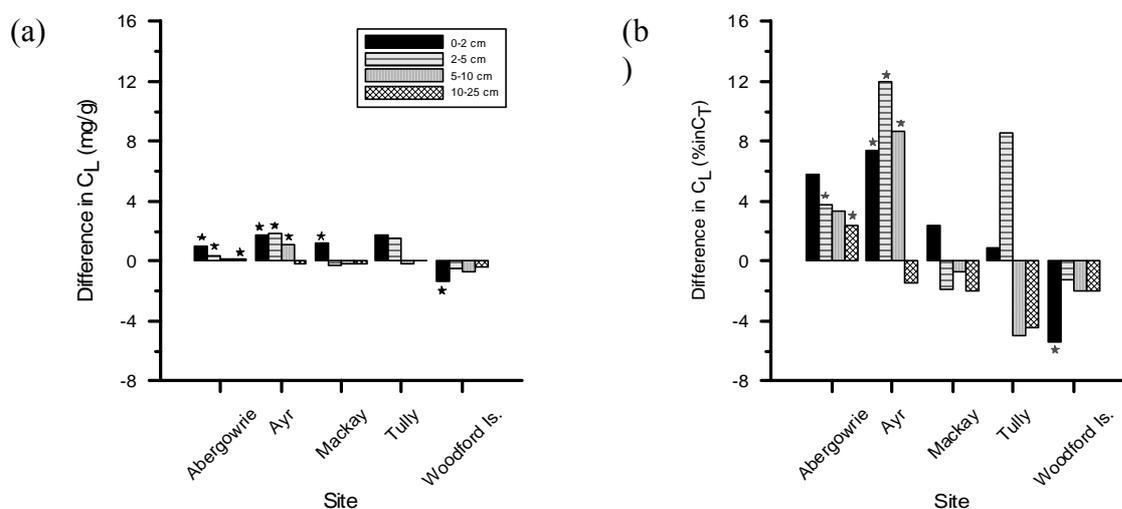


Figure 26. The differences (GCTB-burnt) in (a) the concentration of C_L and (b) C_L as a proportion (%) of C_T at the five study sites. * Indicates a significant difference at $p < 0.05$.

The consideration of C_L in terms of its proportion within C_T revealed similar outcomes to that observed for the concentration data, however there were some variations in results (Figures 26b and 27). Unlike the concentration results no difference between treatments was observed at the Mackay site. Additionally, the depth at which significant ($p < 0.05$) differences were recorded at Abergowrie altered, occurring in the 2-5 cm and 10-25 cm depths only (Figure 26b). Where these variations in significant differences occurred indicates that while trash retention had significantly increased the labile C content it had not resulted in a significant alteration in the composition of the total soil C content between the two treatments.

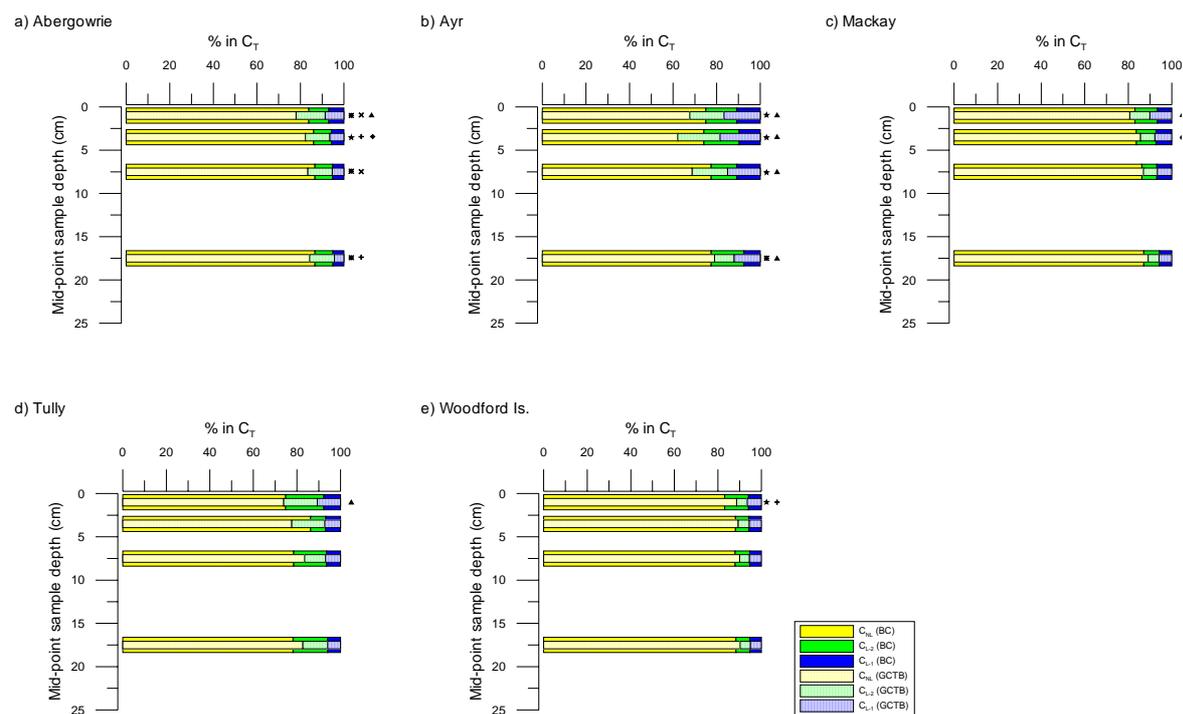


Figure 27. Total C content for the five sites divided into three different fractions based on their readiness to oxidise in two different concentrations of $KMnO_4$. Both treatments are displayed, with the burnt being the soil bar and the GCTB treatment the muted. Symbols \blacktriangle , \blacktriangleleft , \blacksquare , \square , \triangleleft , indicate significant differences at $p < 0.05$, while symbols \blacksquare , \square , \triangleleft indicate significant differences at $p < 0.1$, for the CL1, CL2, and CNL fractions respectively.

The C oxidised by the lower $KMnO_4$ concentration, C_{L1} , demonstrated the same trend for higher concentrations within the GCTB treatment compared to the burnt treatment, but to a greater extent than that displayed by the C_L fraction. This trend occurred at all sites, largely in the surface soil layer, with the treatment differences being significant ($p < 0.05$) in the top 2 cm of soil at Abergowrie and Mackay, to 5 cm at Tully, and to 10 cm at Ayr (Figure 28a). While an increase in C_{L1} was observed at Woodford Island it was not found to be significant. Like that stated above, this trend emphasizes the ability of trash retention to input more labile C into the system than cane burning. In addition, the results further demonstrated that the composition of that input consists of a greater concentration of the more readily oxidisable form of labile C (represented by C_{L1}) and, due to its increased concentration at all sites (especially Woodford Island), that it is likely to be added to the system soon after conversion to trash retention management.

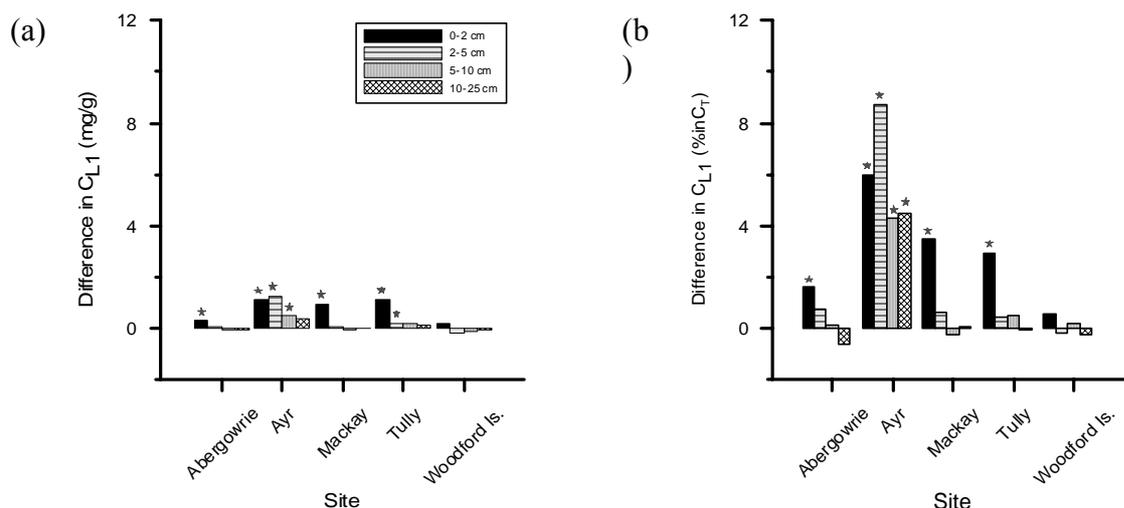


Figure 28. The differences (GCTB-burnt) in (a) the concentration of CL1 and (b) CL1 as a proportion (%) of CT at the five study sites. * Indicates a significant difference at $p < 0.05$.

The proportion of C_T that occurred as C_{L1} again demonstrated this trend for greater amounts within the GCTB treatment compared to the burnt (Figure 28b). The depths to which significant ($p < 0.05$) proportional increases of C_{L1} were observed were the same at the Abergowrie and Mackay sites (to 2 cm), reduced to only the top 2 cm at Tully, and increased to 25 cm at Ayr. The proportion of C_T as C_{L1} at Woodford Is. was higher in the GCTB treatment, but like that reported for the C_{L1} concentration, the treatment difference was not found to be significant. These significant differences indicate that the composition of C_T differs between the two treatments, with trash retention increasing the proportion of C_{L1} within total C, largely in the surface soil layer. The Ayr results indicate that this compositional difference occurred to a greater soil depth, including all soil layers analysed, which may be due to soil type and management (i.e. irrigation) differences. The occurrence of a greater number of significant treatment differences in the proportion of C_{L1} in C_T compared to C_L indicates that trash retention had a larger influence on the enhancement of this most readily oxidisable sub-pool of C_T .

C_{L2} , the fraction derived as the difference between the C oxidised with 333 mM compared to 33 mM of $KMnO_4$ (Eq. 17), was found to have very few significant treatment differences. At the Abergowrie site the GCTB treatment was found to have significantly ($p < 0.05$) higher C_{L2} concentrations than the burnt treatment in the 0-2, 2-5 and 10-25 cm depth increments (Figure 29a). The Ayr, Mackay and Tully sites followed the trend for higher concentrations in the trash treatment, generally in the surface soil, but at no depth were the treatments significantly different (Figure 29a). Again the Woodford Is. site recorded a contrasting result with there being a significantly ($p < 0.05$) greater concentration of C_{L2} in the burnt cane treatment. The reduction in the treatment differences compared to that observed for the two previous C fractions suggests that the differences in the labile C measured by the 333 mM $KMnO_4$ concentration (C_L) consisted largely of the C that was oxidised by the lower $KMnO_4$ concentration (C_{L1}). Thus it appears that trash retention had a tendency to contribute larger amounts of the more readily available labile C. Additionally, with significant increases in the C_{L2} fraction occurring only at the Abergowrie site, the results may indicate that there is an increased temporal requirement for this C fraction to develop as a result of trash retention.

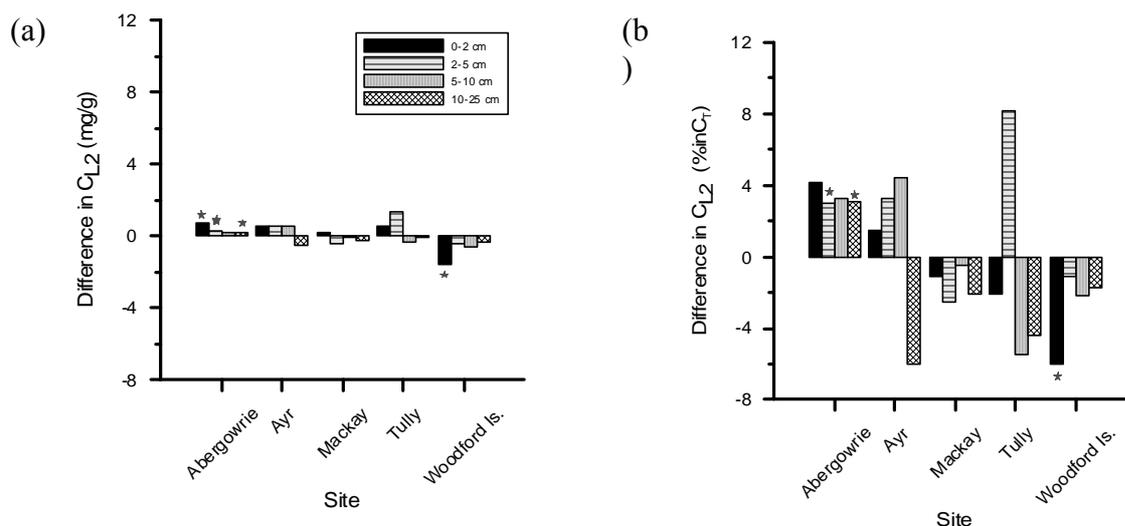


Figure 29. The differences (GCTB-burnt) in (a) the concentration of CL₂ and (b) CL₂ as a proportion (%) of C_T at the five study sites. * Indicates a significant difference at $p < 0.05$.

Little variation between results was observed when considering the CL₂ fraction as a proportion of C_T compared to it as a concentration (Figure 29b). The 0-2 cm soil layer at Abergowrie was the only variation in results, with the depth no longer containing significantly ($p < 0.05$) more CL₂ in the GCTB treatment compared to the burnt (Figure 29b). This would suggest that at this depth the increases in concentration of CL₂ due to trash retention have not resulted in a compositional change of the total soil C compared to the burnt cane treatment. Where significant differences in the proportion of CL₂ in C_T were found indicate that this change had occurred.

The remaining C fraction, C_{NL}, which represented the less oxidisable C or inert C, recorded a general trend for higher concentrations within the surface soil of the GCTB treatment compared to the burnt treatment (Figure 30a). The Mackay and Tully sites both recorded significantly ($p < 0.05$) greater concentrations of C_{NL} within the 0-2 cm soil layer. While Woodford Is., Ayr, and Abergowrie also followed this trend, none were found to be significant and the differences were marginal at the two oldest sites, Abergowrie and Ayr. Below the surface soil layer the concentration of C_{NL} displayed a tendency to be higher in the burnt cane treatment, with both Abergowrie and Ayr recording significant ($p < 0.05$) differences in the 5-10 cm and 2-5 cm soil layers, respectively. These results demonstrate a propensity for trash retention to result in greater concentrations of the less oxidisable C fraction in the surface soil, especially in the cases where GCTB had only been practiced for a medium length of time (e.g. six years or less). The lack of significant treatment differences in the surface soil at the two older sites may indicate that with increased time the additional C_{NL} contributed by trash retention may diminish due to its gradual breakdown. Hence the C that is measured by this fraction is not solely inert C, and instead contains material that requires time to breakdown.

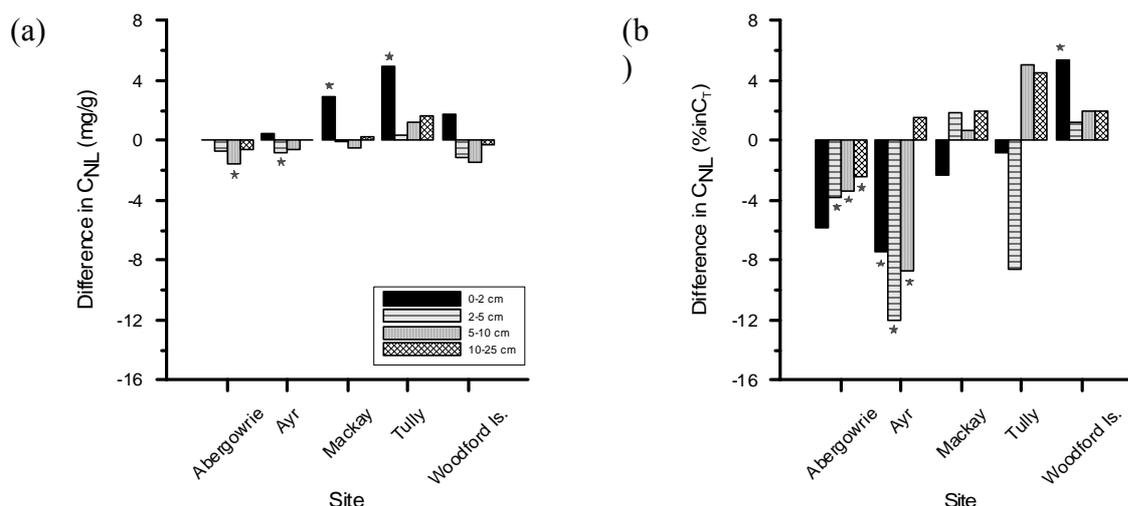
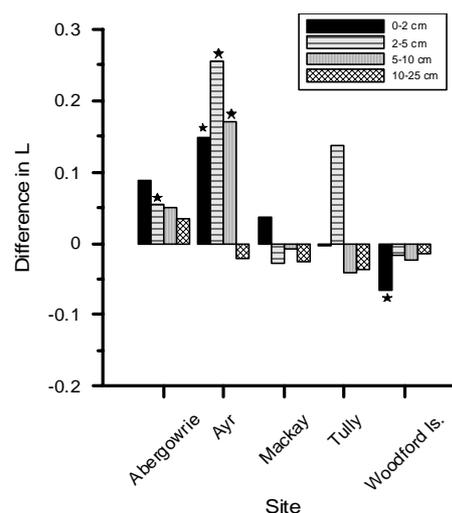


Figure 30. The differences (GCTB-burnt) in (a) the concentration of C_{NL} and (b) C_{NL} as a proportion (%) of C_T at the five study sites. * Indicates a significant difference at $p < 0.05$.

A different perspective of the C_{NL} fraction was observed when the results were considered in terms of the proportion of C_T that occurred as C_{NL} . This method displays a general trend for the proportion of C_{NL} found in C_T to be higher in the burnt treatment compared to the GCTB treatment. This generally occurred in the surface soil layer (0-2 cm) but also at greater depths where trash retention had been practiced for longer time periods (Figure 30b). Abergowrie and Ayr were the only sites where the trend was reflected by significant treatment differences ($p < 0.05$), these occurring at 2-25 cm and 0-10 cm, respectively. The Woodford Is. site reversed this general trend, showing that C_T consisted of a significantly ($p < 0.05$) greater proportion of C_{NL} in the surface soil layer (0-2 cm) of the GCTB treatment than the burnt treatment (Figure 30b). These variations in results compared to that observed for the concentration of C_{NL} indicates that while trash retention may result in an increase in less- or non-labile C forms, there still is a greater proportion of C_{NL} in the system as a result of residue burning.

The results of the lability of C index, L (Eq. 20), which represented the turnover rate of C in the system (Blair *et al.*, 1995), were found to have very few significant treatment differences. Significantly greater L occurred in the upper soil layers of the GCTB treatment at Ayr (to 10 cm) and Abergowrie (2-5 cm) (Figure 31). While the Woodford Is. site had a greater L in the 0-2 cm soil depth for the burnt cane treatment compared to trash retention. There was no difference in lability between trash management treatments at any depth for both the Tully and Mackay sites. The significant results at Abergowrie and Ayr indicate that in those particular soil layers the C turnover rate was occurring at a greater rate in the GCTB treatment compared to the burnt treatment. Ayr was the only site where the C lability was affected by trash retention in more than one soil layer and may be attributed to site specific effects, such as soil type, management, and residue quantity. The higher C turnover rate in the burnt treatment at Woodford Is. may be again accredited to the sites recent change to GCTB management.

Figure 31. The differences (GCTB-burnt) in the index L at the five study sites. * Indicates a significant difference at $p < 0.05$.



UV Photo-oxidation technique

Due to the bulking of replicates prior to testing, no statistical analyses were conducted on the UV photo-oxidation results and hence are reported only as general trends observed in the data.

The results show that the surface soil layers generally contain the greater proportion of the particulate organic C (POC) fraction (Eq. 21). There was a general trend across all sites for the proportion of POC in C_T to decrease with depth and for there to be greater amounts of POC in the GCTB treatment than in the burnt, especially in the surface soil. The POC fraction was higher for the GCTB treatment at Ayr to a depth of 25 cm and at Abergowrie, and Woodford Is. in the surface soil layer (0-2 cm) (Figure 32). Very little difference between treatments was observed at the Tully site (Figure 32). While at Mackay the burnt cane treatment was higher than the GCTB in the upper soil depths (to 5 cm), with POC increasing in the GCTB treatment with depth (Figure 32).

The humic C fraction (Eq. 22) displayed a general trend for higher amounts within the GCTB treatment compared to the burnt treatment in the surface soil layers, with the difference between treatments decreasing with depth. The Mackay site clearly demonstrated this trend, while the Abergowrie, Ayr, and Tully sites displayed some tendencies towards it, especially in the surface soil layers. However, the Woodford Is. site illustrated an inclination for higher humic C in the burnt treatment compared to the GCTB treatment (Figure 32).

As was expected there appeared to be a general trend for a higher degree of inert C in the burnt cane treatment compared to the GCTB treatment (Figure 32). This was especially so in the surface soil layers of the Abergowrie, Ayr, and Tully sites. The Mackay site displayed a tendency to follow this trend but was not as well defined, while at the Woodford Is. site the GCTB treatment contained greater amounts of inert C compared to the burnt treatment. There was also a general trend across treatments and sites for the proportion of inert C with C_T to increase with depth.

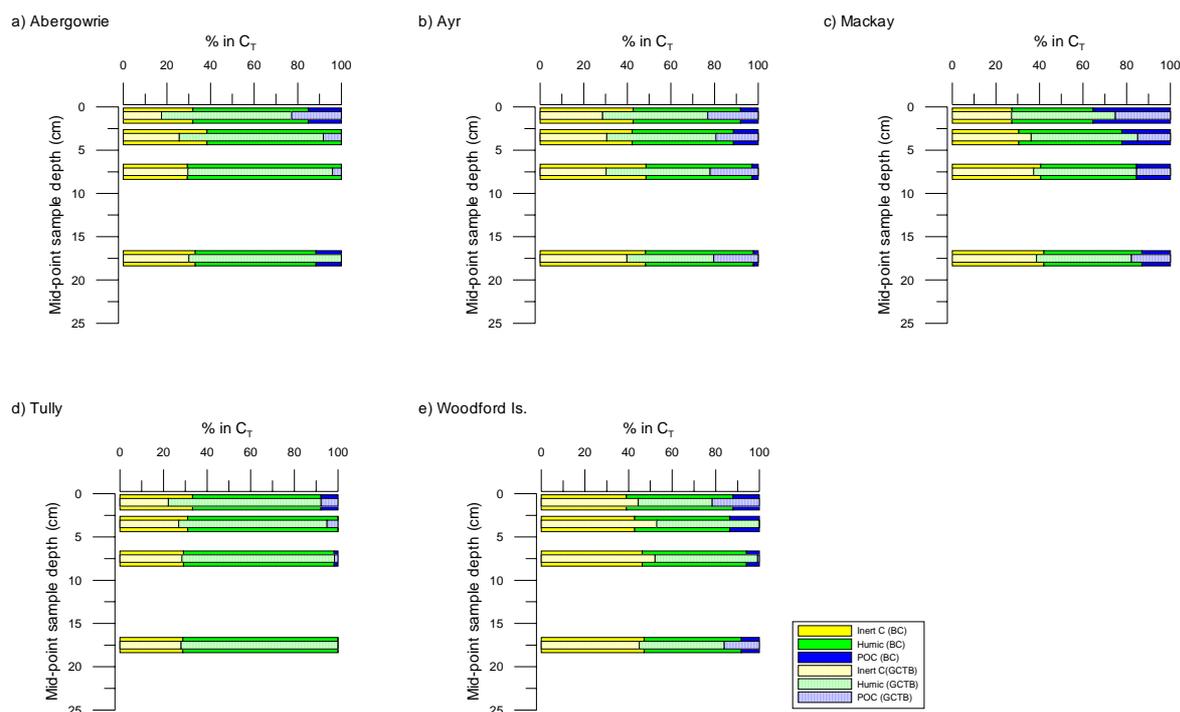


Figure 32. The total C content of the five sites divided into three fractions based on their readiness to oxidise using the second fractionation method. Both treatments are displayed, with the burnt being the soil bar and the GCTB treatment the muted.

The UV photo-oxidation results appeared to have a trend for a greater degree of difference in the inert C fraction between the GCTB and burnt treatments with increased time under the different management practices. This is especially pronounced in the surface soil layers (0-2 cm) with the difference between burnt and GCTB being 14.4%, 14.0%, 11.1%, 0.1%, and – 5.4% (i.e. negative value indicating GTCB>BC), at Abergowrie, Ayr, Tully, Mackay, and Woodford Is., respectively. This mirrors the time lengths under which the sites have been subject to the different management practices (17 yrs, 9 yrs, 6 yrs, 5 yrs and 1 yr, respectively). The other fractions measured by the UV photo-oxidation method did not demonstrate the same tendency to follow the time length under the different management practices.

Correlation Analysis of C Fractions

Scatter plots used prior to correlation analysis suggested that the Woodford Is. data and the lowest soil depth sampled (10-25 cm) should be discounted for the analysis, which occurred for this study.

The correlation analysis conducted on the C fractions determined by the two fractionation methods showed a number of linear associations between the variables. Largely these associations occurred between variables within the same fractionation technique. The KMnO_4 fractions, C_{L1} , C_{L2} , and C_L , were all highly positively correlated ($p < 0.001$) with each other, while the C_{NL} fraction was highly negatively correlated ($p < 0.001$) to these fractions. For the UV photo-oxidation fractions, POC was highly inversely correlated ($p < 0.001$) with both the $<53\text{-}\mu\text{m}$ and humic C fractions; the $<53\text{-}\mu\text{m}$ and humic C fractions were highly positively ($p < 0.001$) correlated to each other, while the humic C fraction was negatively correlated ($p < 0.05$) to the inert C fraction. The details of the correlations between these variables are given in Table 14.

The only linear association of variables across the two-fractionation techniques occurred between the C_{L1} and POC fractions, both of which may be considered as measures of the more readily oxidisable soil C. This was a positive correlation ($p < 0.05$) between these two variables, or a negative correlation if the inverse of POC, the $<53\text{-}\mu\text{m}$ fraction, was considered (Table 14).

Other correlations found occurred in association with total soil C (C_T). C_T was positively correlated to the POC fraction (or negatively if the $<53\text{-}\mu\text{m}$ fraction was considered) and negatively correlated to the inert C fraction (Table 14). Interestingly no correlations occurred between the C_T measurement and the KMnO_4 fractions.

Table 14. Correlation matrix of the C fractions derived through the KMnO_4 oxidation and UV photo-oxidation techniques.

(Values in **bold** indicate those attributes that correlate at $p < 0.05$.)

C Fraction	C_T	C_{L1}	C_L	C_{L2}	C_{NL}	POC	$<53\text{-}\mu\text{m}$ fraction	Humic C
C_{L1}	0.2644							
P value	0.2119							
C_L	0.2529	0.8984						
	0.2331	0.0000						
C_{L2}	0.1989	0.6515	0.9185					
	0.3516	0.0006	0.0000					
C_{NL}	-0.2529	-0.8984	-1.0000	-0.9185				
	0.2331	0.0000	0.0000	0.0000				
POC	0.4458	0.4032	0.2695	0.1026	-0.2695			
	0.0290	0.0508	0.2029	0.6335	0.2029			
$<53\text{-}\mu\text{m}$ fraction	-0.4458	-0.4032	-0.2695	-0.1026	0.2695	-1.0000		
	0.0290	0.0508	0.2029	0.6335	0.2029	0.0000		
Humic C	-0.1076	-0.3777	-0.1772	0.0339	0.1772	-0.7632	0.7632	
	0.6168	0.0688	0.4075	0.8749	0.4075	0.0000	0.0000	
Inert C	-0.4630	0.0063	-0.0902	-0.1615	0.0902	-0.2520	0.2520	-0.4302
	0.0227	0.9767	0.6751	0.4510	0.6751	0.2348	0.2348	0.0359

Nitrogen analysis

Like the C_T results, there was a general trend across all sites, except for Woodford Is., for a significant increase in the N_T content with GCTB compared to the burnt cane treatment. While the trend was similar between the two measures, the depth to which significant differences were observed was greater for N_T . Significantly higher N_T contents were recorded to a depth of 5 cm at Ayr, Mackay, and Tully ($p < 0.05$) (Figure 25b). This was also observed at the Abergowie site, but only when the results were considered at a higher level of significance ($p < 0.1$) (Figure 25b). Woodford Is. did not follow this trend and recorded no significant differences between the two treatments. Additional differences were recorded at the Ayr and Tully sites, with significantly greater N_T contents found to a depth of 25 cm for the GCTB treatment at both sites (Figure 25b). While Abergowie recorded the only significantly higher N_T value for the burnt cane treatment, observed at the 5-10 cm soil depth (Figure 25b).

Like the C_T result, the maximum increase in N_T also occurred at the Tully site in the surface depth of 0-2 cm (increase of 0.006%). Again, like C_T , there was no trend for a proportional increase in N_T content with increased time of GCTB. The increase in N_T contents for the surface soil layer was order the same as for C_T , being Tully (6 yrs)>Mackay (5yrs)>Ayr (9 yrs)>Abergowrie (17 yrs)>Woodford Is. (1 yr), emphasizing the difficulty in extrapolating across sites and through time.

There were no consistent differences between treatments in NO_3-N , NH_4-N , or potential mineralisation (aerobic, waterlogged or biological) for either the Abergowrie or Ayr sites (results not shown).

Other soil chemical properties

As found by Thompson (1966) and Wood (1986), there were very few consistent trends for significant differences in the soil nutrients studied between the GCTB and burnt treatments across all five sites. Significant differences were found between the treatments at various sites and depths but there were minimal trends for treatment and/or depths effects across all sites, emphasizing the major role of site variation in determining the outcome of different management systems on soil nutrients.

An exception was that calcium (Ca cmol_c/kg) values were significantly higher in the burnt cane treatment at all sites, except for Abergowrie ($p < 0.05$). The depths at which these higher Ca contents were observed in the burnt treatment varied from only the surface depth (0-2 cm) at Mackay, to multiple lower depths at Ayr (5-25 cm) and Tully (2-25 cm), and all depths at Woodford Is. (0-25 cm) (Table 15). The Abergowrie site reversed this trend, recording a greater Ca (cmol_c/kg) content in the trash treatment at the soil surface (0-2 cm) ($p < 0.05$) (Table 15).

A slight trend was evident for higher potassium values (K cmol_c/kg) in the surface soil layer (0-2 cm) for the burnt treatment. All sites displayed this trend, except for Ayr, however only the Tully and Woodford Is. sites returned significant results (Table 15). This trend for more K in the burnt treatment also occurred in lower soil depths at Abergowrie, Tully and Woodford Is., but none were significant.

Phosphorus (P mg/kg) contents were also found to be higher in the burnt cane treatment compared to the GCTB, at Abergowrie, Ayr and Woodford. Of these sites, Ayr reported significantly higher results for all four depths (to 25 cm), while Abergowrie and Woodford Is. were only significantly higher at the surface depth (0-2 cm) (Table 15). Mackay followed the trend recording a higher P content in the surface soil layer of the burnt treatment but this was not found to be significant. Tully was the only other site to return significantly different P values, with the GCTB treatment containing a significantly greater amount of P to a depth of 5 cm compared to the burnt treatment (Table 15).

Significantly higher sulphur (S mg/kg) contents were found in the GCTB treatment at the Abergowrie, Ayr and Tully sites compared to burnt cane treatment. These sites followed this trend in all depths but significant differences were only found at the surface layer (0-2 cm) at Abergowrie, 5-10 cm at Ayr, and to a depth of 5 cm at Tully (Table 15). In contrast a significantly greater amount of S was observed in the Mackay surface soil (0-2 cm) for the burnt cane treatment (Table 15). The Woodford Is. results were not statistically analysed due to the number of missing replicates.

Table 15. Summary of the differences between treatments (GCTB-burnt) for the soil nutrients analysed. Negative values indicate where the burnt treatment was greater than the GCTB treatment. (Note – the statistical analysis was not conducted on the S content for Woodford Is. and the pH and EC for Mackay due to a high number of missing replicates.)

Chemical property	Soil depth (cm)	Abergowrie	Ayr	Mackay	Tully	Woodford Is.
Ca (cmol _c /kg)	0-2	0.15*	-0.16	-0.59*	0.06	-0.40*
	2-5	0.05	-0.32	0.26	-0.24*	-0.74*
	5-10	-0.00	-0.78*	0.25	-0.32*	-0.62*
	10-25	-0.02	-0.50*	0.18	-0.26*	-0.87*
CEC (cmol _c /kg)	0-2	0.08	0.3	-0.95*	0.81*	-0.74*
	2-5	0.24	0.48	0.17	0.24*	-0.25
	5-10	0.15	-0.02	0.26	0.08	-0.19
	10-25	-0.19	-0.59	0.41	0.19	-0.6
K (cmol _c /kg)	0-2	-0.08	0.01	-0.11	-0.10*	-0.25*
	2-5	-0.02	0.01	0.07	-0.06	-0.07
	5-10	0.00	0.01	0.06	-0.04	-0.13
	10-25	-0.02	0.00	0.03	0.02	-0.10
Mg (cmol _c /kg)	0-2	0.02	0.08	0.04	0.00	-0.57*
	2-5	0.04	0.01	0.25*	-0.08*	-0.06
	5-10	0.02	-0.16*	0.26*	-0.11*	0.06
	10-25	-0.06	-0.04	0.17	-0.05	-0.03
Na (cmol _c /kg)	0-2	-0.00	-0.00	0.01	0.01	-0.02
	2-5	0.00	-0.00	0.01	0.00	0.03
	5-10	-0.00	-0.01	0.01	0.00	0.07
	10-25	-0.00	-0.04*	0.02*	0.00	0.15*
P (mg/kg)	0-2	-35.25*	-23.8*	-5.25	7.67*	-15.58*
	2-5	-20.75	-16.4*	1.0	3.0*	-13.0
	5-10	-16.75	-9.4*	4.5	-0.33	-11.75
	10-25	-7.75	-12.6*	12.0	1.33	-5.25
S (mg/kg)	0-2	1.42*	0.87	-4.0*	7.67*	-27.83
	2-5	2.25	1.11	-1.75	5.67*	4.25
	5-10	2.75	0.95*	1.42	4.0	0.25
	10-25	0.5	0.02	1.0	-1.0	-2.25
pH	0-2	0.31	-0.10	-1.3	-0.3*	-0.35
	2-5	0.37	-0.16	1.12	-0.25*	0.04
	5-10	-2.18	-0.05	-0.22	-0.28*	-0.01
	10-25	-0.07	-0.11	-0.06	-0.27*	0.12
EC (uS/cm)	0-2	15.5*	7.84	-	6.1*	-72.13*
	2-5	-28.05	8.9	-35.95	0.87	-5.9
	5-10	-2.15	3.16	-0.83	-0.57	-16.18
	10-25	8.85	1.48	-1.78	3.27	-16.75

* Significant at $p < 0.05$.

Other significant differences occur between the treatments at various depths and sites though little can be drawn from these in terms of trends. Refer to Table 15 for a further summary of the differences between the two treatments, and the significant results.

Discussion

Like the variations in results already expressed in the present literature (e.g. Prasad and Power, 1991; Schomberg *et al.*, 1994), the differences found in this study between total soil C and N across sites emphasizes the temporal and spatial heterogeneity of the situation and how numerous factors have an influence in the final outcome (e.g. differences in climate, soil type, cultural practices, and residue quantity and quality). This is demonstrated by the lack of any general relationships between the length of time GCTB management had occurred at a

site and the changes in total C, N and other soil nutrients. Because of this diversity of outcomes, extrapolations across sites and through time are difficult to make. Meaning that many of the answers to the questions that are typically asked by sugar cane growers (such as, when and by how much should N fertiliser management change with the adoption of GCTB?) may be largely site specific.

While temporal trends were not a significant feature in this comparison between management regimes due to site specificity, the general effects of trash retention on the system could be determined. As shown by previous research (e.g. Johnston, 1986; Ladd *et al.*, 1994; Blair *et al.*, 1998) trash retention resulted in a general increase in the total soil C content, largely in the surface soil. Further understanding of this change in soil C was achieved through the use of the oxidation techniques, observing how trash retention affected the C sub-pools compared to a regime of residue burning. Like the findings of other researchers (e.g. Chan *et al.*, 2001; Blair *et al.*, 1995; Moody *et al.*, 1997), the results generally demonstrated that the C fractions derived through the oxidation methods were better indicators of changes in soil C due to trash retention than measures of total soil C.

Previous research has demonstrated that the use of lower concentrations of KMnO_4 (33 and 167 mM) resulted in the reduction of all C fractions at a similar magnitude, with the decline in each fraction being greater than that in the unoxidised fraction (Lefroy *et al.*, 1993; Blair *et al.*, 1995). Thus studies have since concentrated on the C fraction oxidised by 333 mM of KMnO_4 (Blair *et al.*, 1995). However, the results of this study suggest that the use of both the 33 mM and 333 mM concentrations may enable a greater understanding of the changes in C sub-pools due to trash retention. Generally, across all sites the results presented a larger degree of significant differences between treatments in the soil organic C levels at the lower KMnO_4 concentration (C_{L1}) compared to that detected by the 333mM concentration (C_L). Trash retention significantly increased the C_{L1} content and altered the proportion of C_T that occurred as C_{L1} in the surface soil layer at all sites, except for Woodford Is. In contrast the C_L sub-pool did not display a similar sweeping trend across all sites, instead altering the proportion of C_L within C_T only at the Abergowrie and Ayr sites. As these two sites are where GCTB had been conducted the longest, one possible explanation for this result is that under GCTB it may take a considerable length of time for the C_L fraction of C_T to build up, whereas the C_{L1} content alters more rapidly with trash retention. Bell *et al.* (1999) showed that the C_{L1} fraction has a role in aggregate stability which may be one possible contributing factor for the more rapid accumulation of this fraction compared to C_L , as it may be more “attracted” to soil colloids thus the first fraction retained by soil. However, it could also be a function of the ‘quality’ of C in trash blankets. Sugarcane has high concentrations of soluble C (Figure 17) and so trash blankets can have a lot of soluble C. Soluble C would probably provide a direct boost to the C_{L1} fraction.

The effect of trash retention on the C fractions was largely confined to the surface soil layers, except at Ayr and Abergowrie where the effects were observed to greater depths. At Ayr the effects were observed to 10 or 25 cm, which could be attributed to site specific characteristics, namely soil type and/or irrigation usage. For example, the low clay content at the Ayr site implies a less adsorptive capability of the soil which, when combined with irrigation, has likely resulted in the translocation of organic matter down the soil profile causing the observed treatment differences in the C_{L1} , C_L and C_{NL} fractions at the greater depths. Other research has also observed the presence of the labile C component of C_T further down the profile, concluding that the fraction is mobile, and may represent the C source required to drive denitrification and methanogenesis processes (Blair *et al.*, 1995).

10. VERIFICATION OF MODELLING THE IMPACTS OF TRASH MANAGEMENT ON NITROGEN DYNAMICS

Introduction

There are no experiments in sugarcane systems where many, or in most cases any of the terms of the N balance (e.g., mineral N dynamics, denitrification, N leached, N taken up by the crop, etc), have been measured in response to trash management, particularly over the longer-term (e.g., decades). In lieu of such data, we have sought to test the APSIM-Sugarcane model on the limited data that is available, such as sugarcane yields and soil C, in long-term trash management experiments to test if the model would be useful in exploring all the terms in the N balance over long time periods. While variations in sugarcane yields and soil C through time obviously do not completely capture N dynamics of a cropping system, they, along with N fertiliser additions, represent the major sources on N in a cropping system and so the major terms of the N balance. Thus it is a necessary step in modelling N dynamics to be able to simulate these parameters.

The study focused on four trash management experiments. The model's performance was assessed against measured yields (which control N removal from the site in cane and recycling of N in trash blanketed treatments) in all experiments, and, in three of the experiments, total soil C (which is mechanistically linked to the soil N cycle; Paul and Juma, 1981). Other data that could be used to verify the model, such as the amount of trash produced, the N concentrations of the crops, and losses of N to the environment, had not been measured in these experiments.

Methodology

Sites

Two of the experiments were located at the BSES research stations in Mackay (Chapman *et al.*, 2001) and Tully, and had been the subject of other studies in this project. The third study was at Abergowrie in the upper Herbert River valley, and was established by Wood (1986, 1991). The final study was at the South African Sugar Association Experiment Station (SASEX), Mount Edgecombe, South Africa (Van Antwerpen and Meyer, 1998; Graham *et al.*, 1999). The SASEX experiment is referred to as BT1. Details of the Australian experiments are given in Table 11. Long-term average annual rainfall is 950 mm at Mt Edgecombe. The soil at BT1 was a grey, cracking, medium clay. The surface soil pH was approximately 5.5 at all sites. There were eight replicates in the trash blanketed treatment of BT1. At BT1, four varieties had been planted over the course of the experiment; Co281 (1939), Co301 (1948), NCo376 (1957 and 1977) and N16 (1991). Farming operations (e.g. planting, fertilising, harvesting) were performed by hand at BT1.

Model configuration

The APSIM model configuration consisted of modules for soil N and C (APSIM-SoilN; Probert *et al.*, 1998), soil water (APSIM-SoilWat; Probert *et al.*, 1998) and sugarcane residue (APSIM-Residue; with the modifications described above) dynamics, and sugarcane growth (APSIM-Sugarcane; Keating *et al.*, 1999). The modules are one-dimensional, use a daily time-step and are driven by climatic data. The dynamics of water, N, C and roots are simulated in soil layers, with water (and associated nitrate) moving between layers where

gradients exist. The soil organic matter is divided into three “pools”, with FOM representing the fresh organic matter (i.e., roots and incorporated plant residues), BIOM representing the active biomass in the soil, and HUM representing the humified material. Part of the HUM pool is considered inert. The soil water module is a “cascading bucket” water balance model, with water between the drained upper limit (DUL) and saturation draining to the layer below. The drainage rate is controlled by the parameter SWCON, which was set to 0.4 in all layers. The lower limit of plant available water is defined by the parameter ll15. Evaporation from the soil follows Ritchie’s (1972) two-stage evaporation model. The presence of plant residues on the soil surface affects runoff (and hence infiltration) and evaporation. The sugarcane module uses intercepted radiation to produce assimilates, which are partitioned into leaf, structural stalk, roots and sugar. The processes represented in the module are responsive to radiation and temperature, as well as water and N supply. Farming operations (such as fertilisation, planting, incorporation of crop residues through cultivation, or burning of crop residues) can be specified through the MANAGER module, to represent actual or hypothetical conditions.

Parameterisation

Abergowrie, Mackay and Tully

Model parameters were based, wherever possible, on measured data. The soil was divided into seven soil layers, with a total depth of 1.5 m. The initial soil organic C was set equal to values measured at the sites at the beginning of the experiments. Soil C:N ratio was set from measurements of total N and C made at the sites (in 1997-1998) in other parts of this project. Relevant data on the soil organic matter fractions were not available, so these were set equal to values used in previous APSIM simulations of soil N dynamics (Probert *et al.*, 1998), except for the top two layers. In these layers, the BIOM fraction was increased and the fraction of inert organic matter decreased to allow for the higher likely N mineralisation capacity of the tropical soil. The soil water parameters (bulk density, saturation, DUL and ll15) were derived from measurements made in 1998 in other activities in this project. Default parameter values for the relevant sugarcane varieties were used in the sugarcane model. Records of the crop and N management regime employed in the experiments were used to specify farming operations in the APSIM-Manager module. After harvest of each crop, trash weights were reduced by 95 % (based on the measurements made in other parts of this project) in the burnt treatment to simulate pre- and post-harvest trash burning. Daily climate data for the Mackay and Tully experiments were obtained from BSES measurements and Bureau of Meteorology (BOM) records for the closest BOM station. The BOM records were used to fill in missing data in the BSES records. For Abergowrie, patched point data from the BOM and Silo web sites were used.

BT1

As with the Australian experiments, soil model parameters and management variables were based on the experimental data for the site obtained from SASEX records. The soil was divided into seven soil layers, with a total depth of 1.5 m. Starting soil C contents in the APSIM-SoilN module were taken from total soil C measurements made prior to the experiment’s commencement (J.H. Meyer, unpublished data). The C:N ratio of the soil was estimated from the average (across treatments and through time) of measurements made during the latter stages of the experiment. Soil water parameters were determined from soil moisture characteristic data from the site (R. Van Antwerpen, unpublished data), measurements of runoff from a similar soil under a range of crop residue treatments in central Queensland, Australia (Thorburn, 1992) and default values for cracking clays given by Probert *et al.* (1998). Parameter values for NCo376 (Keating *et al.*, 1999) were used for simulating

sugarcane in the experiment, as parameters for the other varieties were not available. Daily climate data were obtained from the SASEX records. Prior to 1960, daily measurements of solar radiation were not available. For these years, the mean daily radiation for the years 1960-1999 was used. Simulations were conducted for the period 1939-2000.

Management data (planting and harvesting dates, type, amount and date of N fertiliser applications) were obtained from SASEX records, as were cane yields. Fertiliser was spread on the soil surface in the experiment, so there would have been loss of N from volatilisation of ammonia in the years when urea was used as the N source (from 1973), particularly in the trash blanketed treatment. These losses were estimated for both treatments using the model of Schumann (2000) and the N applications reduced accordingly. As with the other sites, trash weights were reduced by 95 % in the burnt treatment after harvest of each crop to simulate pre- and post-harvest trash burning.

Results

Abergowrie

There was generally good agreement between the measured and simulated yields in both treatments (Figure 33a), but with a tendency to over predict yields in the burnt treatment in some years. This resulted in the mean simulated yield in the burnt treatment being 7 t/ha higher than the GCTB treatment, whereas it was 3 t/ha lower in the trial results. The higher simulated yields in the burnt treatment may have been due to factors such as weed competition, pest damage or lodging, which are not considered in the model, affecting the actual yields.

Increased soil C in the GCTB treatment relative to the burnt treatments was well predicted by the model (Figure 33b). Presumably, the year-to-year variation in the measured differences reflects sampling and analytical variations.

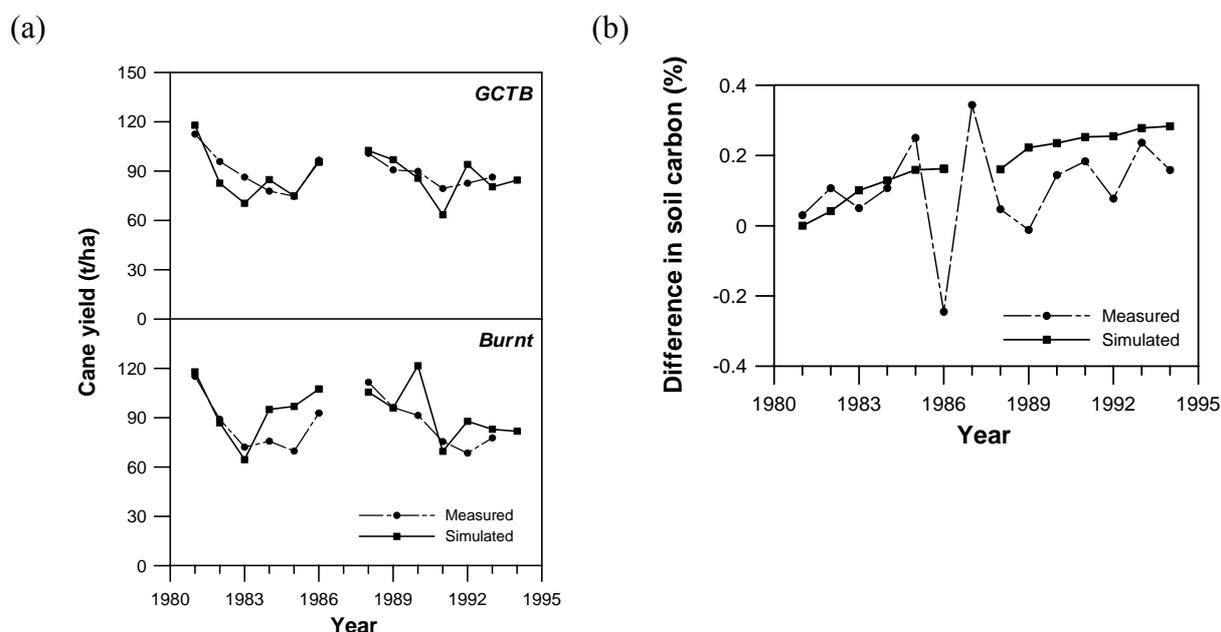


Figure 33. Measured and simulated (a) sugarcane yields under two trash management treatments, and (b) differences in total soil carbon between GCTB and burnt trash management treatments over 0-0.2 m soil depth at the Abergowrie site.

Mackay

Simulated cane yields agreed well with those measured in the experiment at Mackay for all crops except the fifth ratoon harvested in 1999 (Figure 34a). The stools of the fifth ratoon crop were in very poor condition with many gaps, and this reduced the yields. These conditions are not represented in APSIM-Sugar, and resulted in predicted cane yields being higher than those measured. The mean difference between the trash blanketed and burnt treatments measured in the experiment was 8.6 t/ha, compared with a simulated difference of 7.4 t/ha. The simulations indicate that this yield increase in the trash blanket treatment was due to greater water availability in the trash blanketed treatment.

Simulated total C agreed well with measured total C in the experiment (Figure 34b). The total C contents in the burnt and trash blanketed treatments were not significantly different (over the 0-0.25 m depth) from measurements made in 1998 (Figure 25), and the simulated difference between the two treatments was less than the 95 % confidence interval of the experimental means.

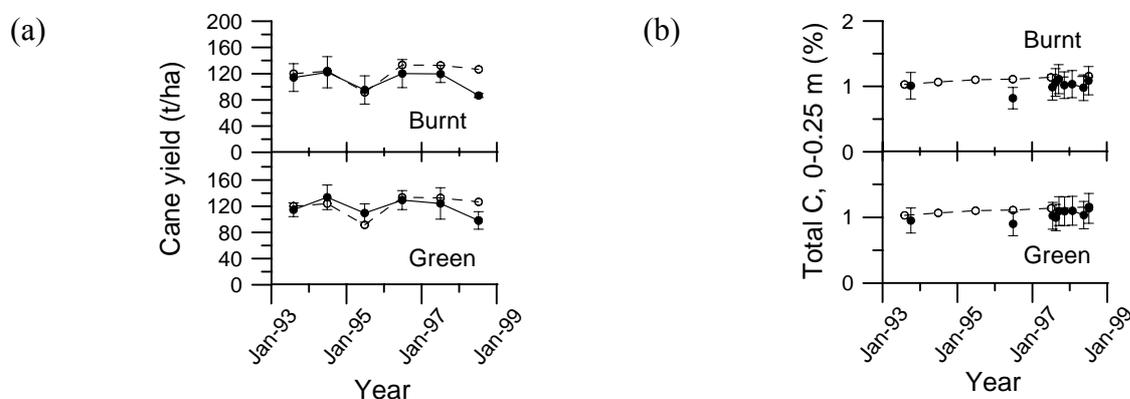


Figure 34. Measured and simulated (dashed lines) (a) cane yield and (b) total soil carbon in burnt and trash blanketed (Green) treatments of the Mackay trash management experiment. Bars about the measured data points show the $\pm 95\%$ confidence interval.

BT1

In the BT1 experiment, predicted cane yields generally agreed with measured yields (Figure 35a), with the mean predicted yield less than 2 t/ha different from the mean measured yield over the whole experiment. Simulated yields were markedly different from measured yields in both treatments in 1956, 1987 and 1990, and in the burnt treatment in 1974, 1976 and 2000. Management records for the experiment did not indicate any apparent reason for these differences. The mean measured cane yield in the trash blanketed treatment was 13.9 t/ha greater than in the burnt treatment, with this difference being 11.5 t/ha for the simulated yields. The simulations indicate that greater water availability in the trash blanketed treatment was a significant factor in this yield increase.

Total C measured in studies conducted since 1980 is well predicted by the model, while there are substantial differences between measured and predicted values in 1945 and 1962 (Figure 35b). It is possible that these differences may be due to methodological factors. The difference between the burnt and trash blanketed treatments is well predicted, even for these early studies. The simulations of total C indicate that soil organic matter has declined in both

treatments of the BT1 experiment. The rate of this run down was greater in the burnt treatment, but became negligible in the trash blanketed treatment during the latter part of the experiment (i.e., the 1990s.).

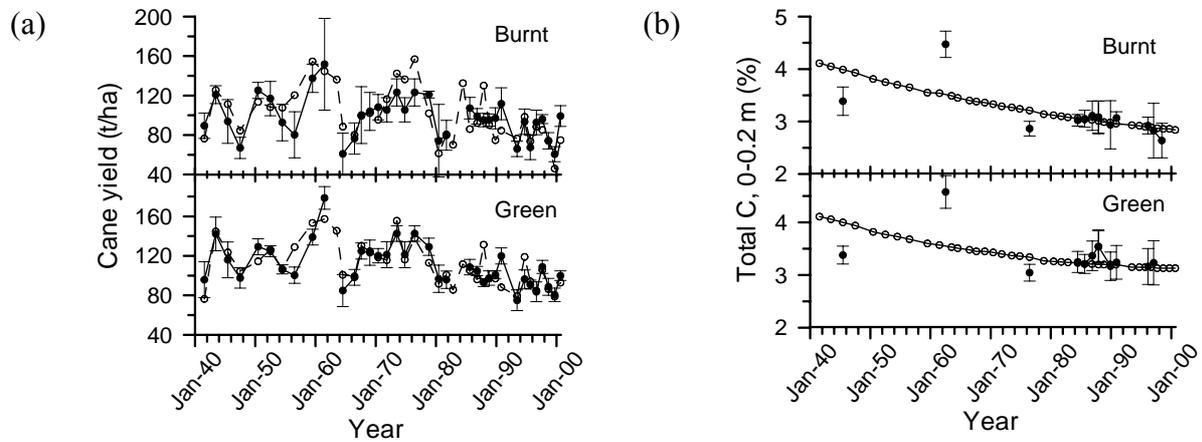
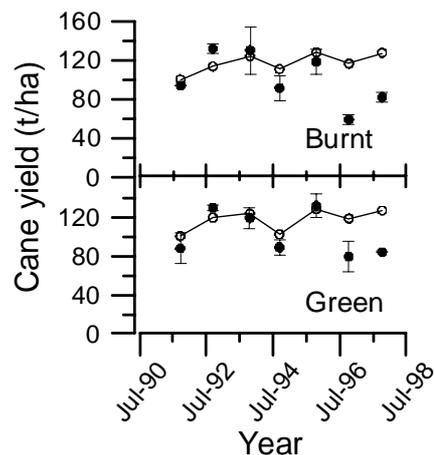


Figure 35. Measured (solid symbols) and simulated (dashed lines – open symbols) (a) cane yield and (b) total soil organic carbon in burnt and trash blanketed (Green) treatments of the BT1 trash management experiment. Bars about the measured data points show the \pm 95 % confidence interval.

Tully

As with the other sites, cane yields were generally well predicted at Tully for the first four crops (Figure 36). As with Mackay, yields in the later (i.e., final two) crops were over predicted. The yield of these crops was much lower than earlier crops due to conditions not modelled (such as harvest damage).

Figure 36. Measured (solid symbols) and simulated (dashed lines – open symbols) cane yield in burnt and trash blanketed (green) treatments of the Tully trash management experiment. Bars about the measured data points show the \pm 95 % confidence interval.



Discussion

The sugarcane cropping system has features (such as the long time over which crops grow, production of high amounts of low quality crop residue, the infrequency of fallows in the cropping system, and the need for large amounts of N to reach potential yields) that set it

apart from many other cropping systems. These features place special demands on a system for modelling N dynamics in a cropping system. To date, only APSIM-Sugarcane has these features. The comparisons of the predicted dynamics of yield, crop residue, and soil N and C with those measured in a range of trash management experiments in the world (Figures 33-36) demonstrates that APSIM-Sugarcane is capable of predicting many of the important attributes of a sugarcane system in the context of trash management. Thus the model has potential to provide important insights into soil organic matter and N dynamics under different residue management practices and optimising the management of N fertiliser where residues are retained.

There are undoubtedly further refinements that could be made to improve the model's performance for these scenarios. More importantly, however, there are processes that are not represented in the model that are required to better address questions such as gaseous losses of N and the impact of N on sucrose yield. Some of these can be partly overcome by use of other models to refine inputs to or outputs from the model (e.g. accounting for volatilisation of N from urea fertiliser as done when simulating the BT1 experiment). The detail required to incorporate these processes into the model may be too great in relation to the other crop, soil and management processes represented in the model.

11. THE IMPACT OF TRASH RETENTION ON K REQUIREMENTS FOR SUGARCANE

Introduction

There is a limited understanding of the short-medium term dynamics of K release from trash blankets, and the availability / stratification of K released for trash. The following questions are central to understanding how K should be managed in trash blanketed crops:

- What is the residual value of K fertilizer
- What is the availability of GCTB-K to subsequent crops
- Does GCTB affect optimal rate of K fertiliser?
- Does GCTB affect stratification of K in the soil?

Answering these questions requires a combination of field and more controlled pot experiments. Field experiments can directly test the agronomic impacts of trash and K fertiliser management interactions. However, their results are likely to be complicated by factors such as variability in deposition of trash by commercial harvesters and difficulties associated with management of water regimes. Pot experiments are able to study processes such as K release from trash, and its stratification and in the soil and subsequent availability to crops. The pot trials will thus help establish if GCTB-K is broadly equivalent to fertiliser K, and help quantify the potential supply from GCTB adoption.

In this section, the experiments undertaken to understand K dynamics in trash blanket systems are described. The implications for management of K fertiliser in trash blanketed systems are discussed.

Methodology

Field experiment

To determine if trash blanketed sugarcane crops could be grown with less K fertiliser than burnt crops, a field experiment was established on a commercial farm in 1996. The farm (Mario Leotta's) was in the Marian mill area of Mackay region, central Queensland. Sugarcane (var. Q124) was conventionally planted during October 1996 (Table 16) and grown in a sandy loam (Kurosol) soil that was likely to have low K fertility. The site had been regularly burnt in the years preceding this. Management of the site was uniform during the plant crop.

Table 16. Summary of operations at the experimental site.

Year	Operations	Comment
1996	Planted to Q124	Uniform management
1997	Plant crop harvested, treatments imposed	
1998	Nil	1R crop stood over
1999	1R crop harvested, treatments imposed	
2000	2R crop harvested. Treatments imposed with trash being incorporated in to the soil in the trash retained plots	Crop badly affected by oranges rust
2000	Q124 ploughed out, then planted to Q138 and fertiliser treatments imposed.	
2001	Q138 plant crop harvested	Crop badly affected by water stress

Following harvest of the plant crop in October 1997, K fertiliser application and trash management treatments were imposed on the site. There were five different K fertiliser applications rates (0-160 kg ha⁻¹ K) and the trash was either removed (to be equivalent to a burning treatment), or retained *in-situ* (at a rate equivalent to 9.2 t DM ha⁻¹ in 1997) to create a green cane trash blanket (GCTB). The experimental design was therefore a two-factor randomised block design, with three replicated blocks.

The first ratoon crop could not be harvested in 1998 due to exceptional weather conditions and the crop was stood over until July 1999 (Table 16). The second ratoon crop, harvested in July 2000, was badly affected by orange rust. Following that harvest, the trash was removed from the relevant plots as usual, but trash was incorporated into the soil in the trash retained treatment. Q124 crop was then ploughed out and replanted with variety Q138 in October 2000. It was thought that incorporation of the trash was the best way to retain the K contained in the trash blanket within the trash retained plots through the plough out-planting operation, and allow the experiment to continue for the 2000-2001 year. The Q138 plant crop was harvested in October 2001, after a year of low rainfall and widespread water stress in the Mackay district.

At each harvest, the yield and CCS of the sugarcane was determined. The K concentration in the cane was also measured. In 1999, the mass and K concentration of the trash blanket was measured. However, since K is easily leached from trash, it is possible for treatment effects to be obscured by rain between harvest and trash blanket sampling. Thus, in 2000 trash (tops and dead leaves) was sampled directly from plants (15 stalks) in the plots immediately prior to harvest and determine K concentration of this material.

Pot experiments

A series of five pot experiments were conducted in a poly-tunnel shelter at BSES Mackay. The first three experiments used the K-rich illitic soil (Chromosol) from the long term K trial on the Mackay SES. The fourth experiment also used the K-rich illitic soil, but from the fallow block of D2 at Mackay BSES. The fifth experiment used a sandier and lower K (Kurosol) soil from the Leotta K rate trial at Mirani. Soil data relevant to establishment of experiments with soil from the above sites are summarised in Table 17.

Experiment 1: K response on soils of differing K inputs

This experiment was established to quantify the residual value of K applied historically to this soil for 32 years. Fresh K additions were added to a range of historically fertilised treatments to allow trends in differences to be identified.

K fertiliser treatments (0, 56, 112, 140, 168 and 224 kg K/ha) were applied to four replicate pots of 38cm diameter containing 18 kg of air dried soil from selected histories of K fertiliser inputs (0, 28, 84, 168 and 196 kg ha⁻¹ K). Soil was contained within a pot with out holes to prevent K loss from the system. The variety Q138 was planted as germinated one-eye setts, three to each pot prior to the application of fertiliser. The surface of the soil in the pots was covered with river stone to (1) act as mulch and limit evaporation in Experiment 1 and (2) to separate trash from the soil in the second and third experiment.

The whole plant was harvested to ground level 168 days after planting. Whole plants were weighed (field moist), put through a cutter grinder and sub sampled for moisture (dried at 80°C). Dry plant samples were then analysed for (Ca, Mg, P, K, S, Cu, Zn, Fe, Mn, Al). The soil was immediately sampled with a pogo stick sampler soil sampler, air dried and put through a 2 mm sieve prior to analysis. The assays included (pH, EC, cations, P, S and nitric K).

Experiment 2: Effect of cane trash on soils with differing history of K inputs

Four replicate pots of each soil derived from plots with differential K input (0, 28, 56, 84, 112, 140, 168 and 196 kg K/ha) were established before river stone and four rates of chopped up trash (0, 5, 10 and 20 t DM/ha), were added (Table 18). This made the experiment a complete two-factor design with four fully randomised blocks.

The variety Q138 was planted in this experiment in line with procedures used in experiment 1. The whole plant was harvested to ground level 175 days after planting, and trash was collected. Whole plants and trash were weighed separately (field moist), put through a cutter grinder and sub sampled for moisture and chemical analysis. The plant and trash samples were then analysed for (Ca, Mg, P, K, S, Cu, Zn, Fe, Mn, Al). The soil was immediately sampled with a pogo stick soil sampler, air dried and put through a 2 mm sieve prior to analysis. The assays included (pH, EC, cations, P, S, nitric K).

All pots from experiments one and two were irrigated at the same time using an irrigation system with radial nozzles when tensiometers read about 80 centibars of suction. The amount of water applied was recorded for each irrigation event.

Experiment 3: Availability of trash-K to sugarcane

Standard sugarcane trash was collected from a conventionally fertilised crop, chopped up and air dried. These were then added to four replicate pots of soil that had previously received 112 kg K/ha in the long-term K trial. River stone was used to separate the trash from the soil. The trash was added at 10 t DM/ha. Twenty additional pots were set up to allow destructive sampling after approximately 10, 25, 40, 62.5, and 115 mm of simulated rainfall (deionised water). These 'rainfalls' were added at the same time and in single events (Table 19). The pots had a radius of 19 cm, which made 1134 ml equivalent to 10 mm of water. The final destructive sampling of trash occurred at 104 days after the first water application event.

Table 17. Details of soils used in pot experiments to study interaction with GCTB residues and potassium dynamics at Mackay.

Experiments 1-3									
Soil source : Mackay BSES Long-term K Expt					Soil type: Pioneer, Non-calcic brown (Chromosol)				
Particle size	CS	FS	S	C					
(%)	10	55	16	17					
K history (kg K/ha)	pH	EC (dS/m)	Ca (cmolc/ kg)	Mg (cmolc/k g)	Na (cmolc/kg)	K (cmolc/ kg)	NK (cmolc/ kg)	P (mg/ kg)	S (mg/ kg)
0	5.05	0.039	2.29	0.67	0.10	0.09	1.18	53	6
28	5.12	0.037	2.82	0.81	0.10	0.11	1.42	47	7
56	5.02	0.041	2.25	0.71	0.10	0.12	1.25	53	8
84	5.13	0.038	2.80	0.73	0.10	0.11	1.37	46	6
112	5.14	0.040	3.03	0.83	0.11	0.14	1.47	47	6
140	5.15	0.037	2.65	0.75	0.10	0.15	1.31	52	6
168	5.14	0.038	2.73	0.75	0.10	0.17	1.48	50	6
196	5.09	0.042	2.45	0.74	0.10	0.20	1.46	45	6

Experiment 4									
Soil source : Mackay BSES, Block D2					Soil type: Pioneer, Non-calcic brown (Chromosol)				
Particle size	CS	FS	S	C					
(%)	10	55	16	17					
	pH	EC (dS/ m)	Ca (cmolc/ kg)	Mg (cmolc/ kg)	Na (cmolc/ kg)	K (cmolc/ kg)	NK (cmolc/ kg)	P (mg/ kg)	S (mg/kg)
	5.42	0.030	2.56	1.26	0.11	0.23	1.25	64	12

Experiment 5									
Soil source : Leotta's farm at Mirani					Soil type: Kinchant, Soloth (Kurosol)				
Particle size	CS	FS	S	C					
(%)	39	50	7	10					
	pH	EC (dS/m)	Ca (cmol _c /k g)	Mg (cmol _c /k g)	Na (cmol _c /k g)	K (cmol _c /k g)	NK (cmol _c /k g)	P (mg/k g)	S (mg/kg)
	5.74	0.050	2.31	1.41	0.20	0.10	0.19	8	6

Where, CS = coarse sand, FS = fine sand, S = silt, C = clay and NK = Nitric K.

Table 18. Chemical analysis of dry trash applied to experiments 2, 3, 4 and 5.

Trash	Ca %	Mg %	P %	K %	S %	Cu mg/ kg	Zn mg/ kg	Fe mg/ kg	Mn mg/ kg	Al mg/ kg
Experiment 2	0.309	0.182	0.084	0.781	0.104	2.3	17.4	224	147	146
Experiment 3	0.216	0.136	0.090	0.827	0.100	2.2	15.1	188	192	115
Experiments 4 & 5	0.483	0.154	0.067	0.790	0.100	-	-	-	-	-

Table 19. Volume and timing of cumulative irrigation events for experiment 3.

Treatment (mm)	Water application events (ml/19 cm radius pot):													
	1	2	3	4	5	6	7	8	9	10	11	12	13	14
10	1134													
25	1134	1702												
40	1134	1702	851	300	551									
62.5	1134	1702	851	300	551	851	851	851						
115	1134	1702	851	300	551	851	851	851	851	851	851	851	851	1702
Days after treatment	0	0	8	11	18	50	68	77	84	88	88	95	100	104
Cumulative irrigation (mm)	10	25	32.5	35.2	40	47.5	55	62.5	70	77.5	85	92.5	100	115

The plants were harvested 166 days after the commencement of the experiment and were watered as required throughout. Whole plants and trash were weighed (field moist), put through a cutter grinder and sub sampled for moisture and chemical analysis. Plant and trash samples were then analysed for (Ca, Mg, P, K, S, Cu, Zn, Fe, Mn, Al). The soil was immediately sampled with a pogo stick soil sampler, air dried and put through a 2 mm sieve prior to analysis. The analysis included (pH, EC, cations, P, S and nitric K).

All pots from the above three experiments were fertilised with above recommended rates for N and P to ensure other nutrients did not limit crop growth and K uptake.

Experiment 4 and 5: Assess the fate of K leached from trash

The previous pot experiments were run in a closed system, with the K applied as trash or fertiliser destined for plant uptake, remaining adsorbed on the soil complex or in soil solution. But the closed system resulted in waterlogging of some pots in experiment 1 and particularly in experiment 2, therefore a drainage and leachate collection system was applied to experiments 4 and 5. The main purpose of the latter experiments is to look at stratification of K in the top 20 cm of soil when K is leached from trash and to see if it is leached below the top 20 cm zone.

Standard sugarcane trash was collected from a conventionally fertilised crop (Table 18), air dried in the glasshouse and chopped (5 cm pieces). These were added to three replicate pots of soil from the old GCTB fallow blocks that had a previous history of trash retention (one crop cycle). Mesh was used on the bottom of the pots as well as to separate trash from the soil. Trash was added at 10 t DM/ha. Enough pots were included to allow destructive sampling after 25, 50, 100, 150, and 200mm of simulated rainfall applied in 25 mm events. All pots received a preliminary 25 mm before trash was applied and the soil sampled and leachate collected for an initial assessment. Applied water came from the reticulated supply (Table 20).

Table 20. Analysis of water used to simulate rainfall in experiments 4 and 5.

	Ca mg/L	Mg mg/L	Na mg/L	K mg/L	P mg/L	S mg/L	pH	EC dS/m
Applied water	36.36	23.79	59.40	0.75	0.056	4.90	7.11	0.584

The trash was then removed, dried and weighed. It was analysed for K, Ca, Mg, P and S. Soil was sampled by a pogo stick sampler and broken up into increments 0-2 cm, 2-5 cm, 5-10 cm; and 10-25 cm.

Trash was removed from relevant treatments when the accumulated irrigation target was reached. Leachate volume was measured then filtered and analysed for the same elements as in the soil assay above.

The experiments had three replicates with seven treatments (Table 21). Experiments 4 and 5 are companion experiments with two different soil types (high K soil; low K soil). They were run consecutively. All pots were irrigated before commencement of the experiment to bring soil to near saturation (5 L for experiment 4 and 5.5 L for experiment 5). A simulated rainfall event of 25 mm was equivalent to adding 2835 ml of water.

Table 21. Water management treatments applied to experiments 4 and 5.

Treat No.	Treatment description
1	Control –end; bare soil sampled at end of experiment along with leachate.
2	Control – start; bare soil sampled after preliminary 25 mm leaching, prior to trash.
3	Trash removal after 25 mm, soil sampled, leachate measurements.
4	Trash removal after 50 mm, soil sampled, leachate measurements.
5	Trash removal after 100 mm, soil sampled, leachate measurements.
6	Trash removal after 150 mm, soil sampled, leachate measurements.
7	Trash removal after 200 mm, soil sampled, leachate measurements.

Results

Field experiment

Cane yields of the Q124 plant crop harvested in 1997 were similar across all the plots to which treatments were subsequently applied (Figure 37a). There were no variations in CCS either (data not shown). There were significant ($P < 0.05$) treatment main effects in the 1999 crop. Cane yields increased with increasing K rate and with retention of trash (Figure 37b). Results were similar for the crops harvested in 2000 (Figure 37c) and 2001 (Figure 37d), although the treatment differences were not as great in 2000.

Sugar yield was little affected by the K and trash treatments in 1999 (Figure 38a), although CCS values were very low (mean of 5.2) and variable ($SD = 2.3$). Sugar yields reflected cane yields in 2000 and 2001 with similar treatment responses (Figures 38b and c).

The concentration of K in the trash blanket (following harvest) was not significantly affected by either K application or trash retention in 1999 (Figure 39). However, it is likely that rain which fell between harvest and trash blanket sampling leached K from the trash blanket. In subsequent years, when trash was sampled from the plants prior to harvest, K concentrations significantly increased with increasing rate of applied K (Figures 40a and b), but were only significantly higher when trash was retained in 2000 (Figure 40a).

The concentration of K in the harvested cane significantly increased with increasing rate of applied K in the crops harvested in 1999, 2000 and 2001 (Figures 41a, b and c). The concentration of K in the harvested cane was also significantly higher when trash was retained in 1999 and 2000.

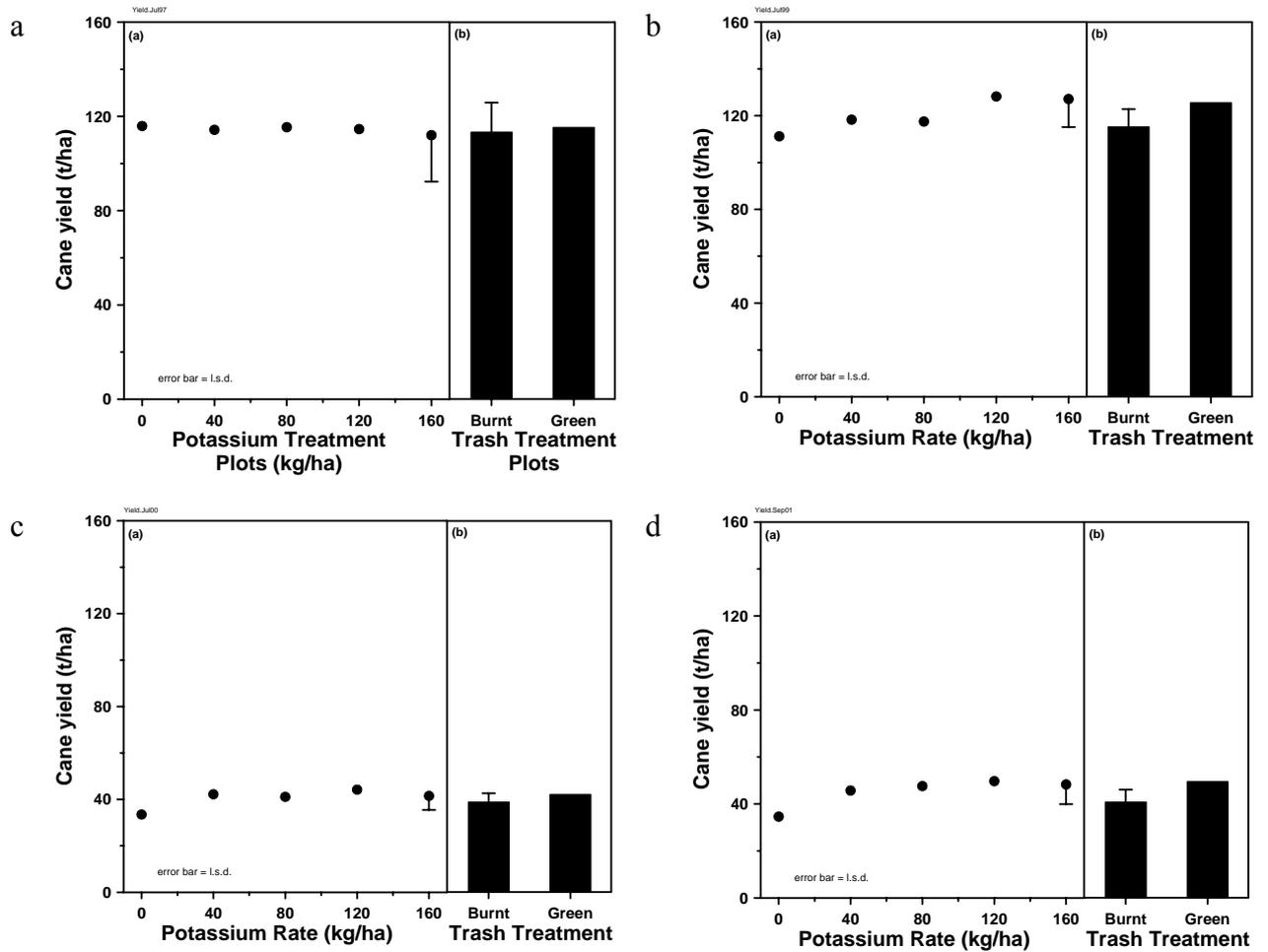


Figure 37. Mean yield of sugarcane for different K rate main effects and trash treatment main effects for crops harvested in (a) 1997, (b) 1999, (c) 2000 and (d) 2001. Error bars show the Lsd (P = 0.05) for the main effects.

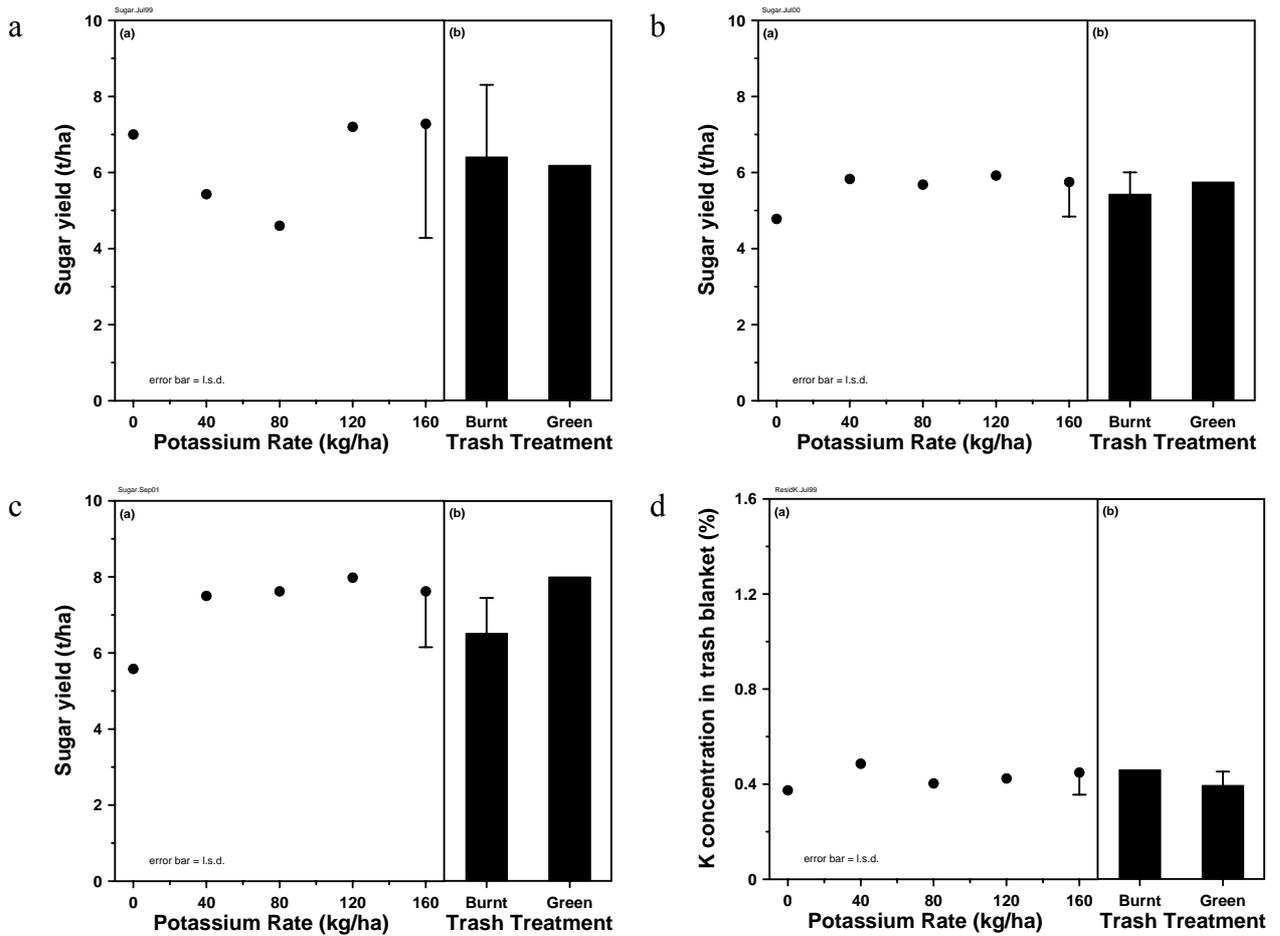


Figure 38. Mean yield of sugar for different K rate main effects and trash treatment main effects for crops harvested in (a) 1997, (b) 1999, (c) 2000 and (d) 2001. Error bars show the lsd (P = 0.05) for the main effects.

Figure 39. Mean K concentration for different K rate main effects and trash treatment main effects of the trash blanket when harvested in 1999. Error bars show the lsd (P = 0.05) for the main effects.

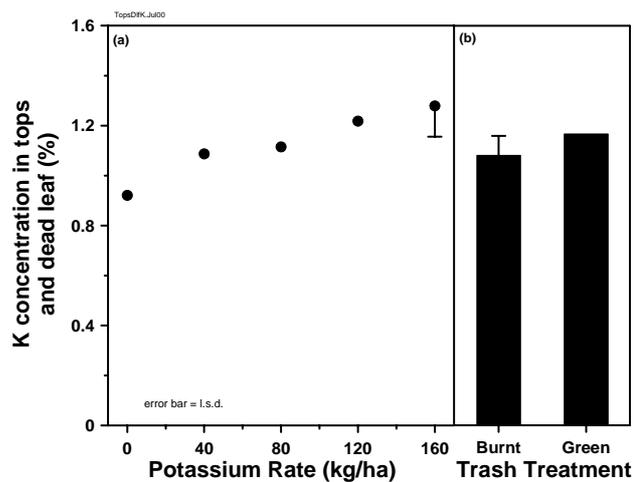


Figure 40. Mean K concentration for different K rate main effects and trash treatment main effects of plants when harvested in 2001. Error bars show the lsd (P = 0.05) for the main effects.

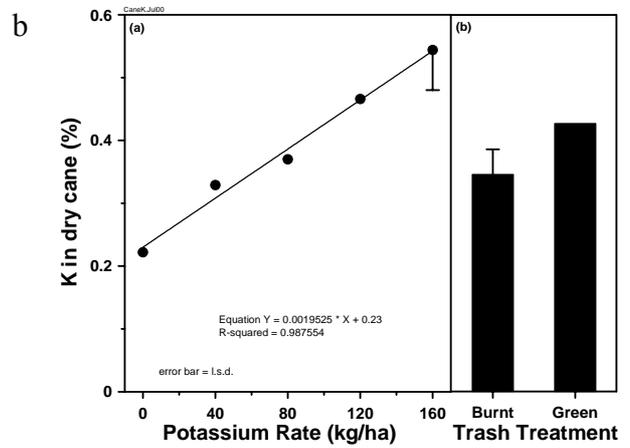
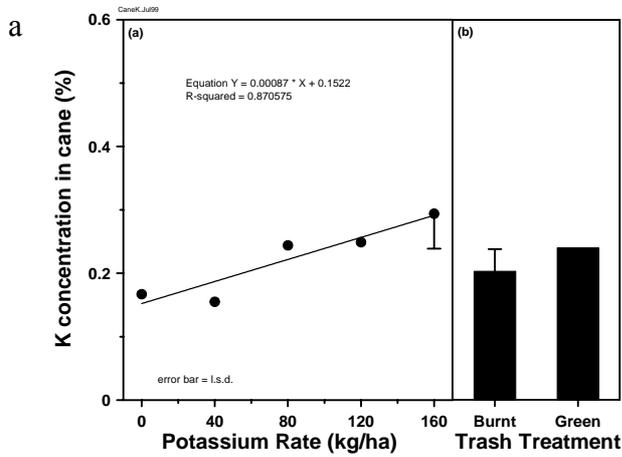
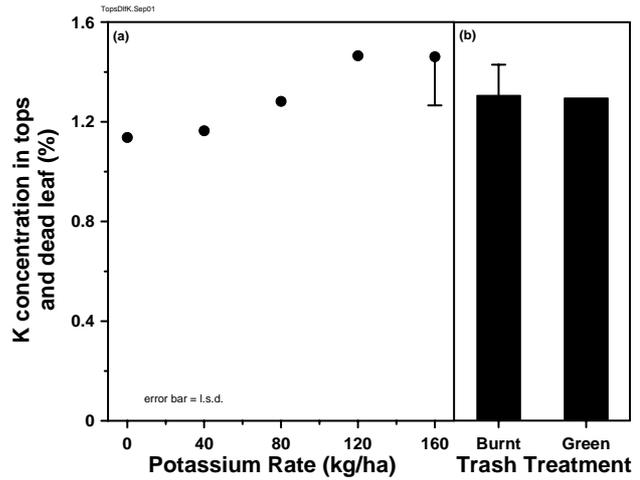
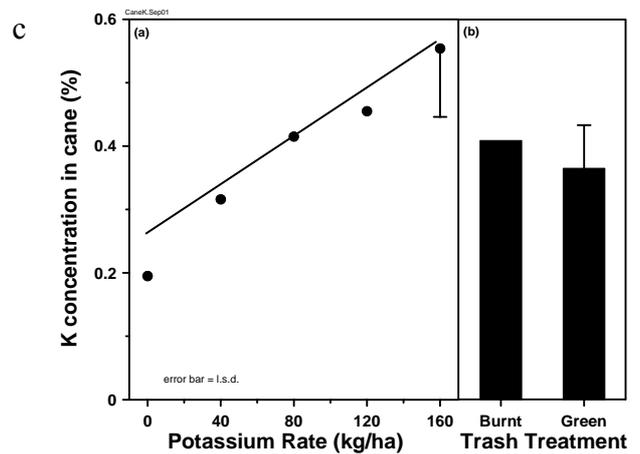


Figure 41. Mean K concentration for different K rate main effects and trash treatment main effects of cane when harvested in (a) 1999, (b) 2000 and (c) 2001. Error bars show the lsd (P = 0.05) for the main effects and the lines indicate the linear regression.



Pot experiments

Experiment 1: K response on soils of differing K inputs

Analysis of variance for biomass yield from experiment 1 showed that applied fertiliser had a significant effect on both fresh and dry matter yield (Table 22). Fresh biomass yield of all fertilised treatments was significantly greater than the unfertilised control treatment, however differences between fertilised yields generally were not significantly different. Only the 224 kg K/ha treatment gave a significantly higher dry matter yield than the control. Analysis of biomass dry matter % data (not shown) revealed that potassium fertilisation was associated with higher moisture content. Values for 112, 168 and 224 kg K/ha treatments were significantly higher ($p < 0.05$) than the 0 kg K/ha control treatment; differences in moisture content between fertiliser treatments were not significant. Previous history of potassium fertilisation and the interaction between potassium rate and history were not significant in terms of yield.

Table 22. Yield of sugarcane biomass at harvest of experiment 1.

Fertiliser K (kg K/ha)	Fresh biomass (g/pot)	Significance of means (FW)*	Dry biomass (g/pot)	Significance of means (DW)
224	1091.7	a	224.9	a
168	1045.5	a b	220.1	a b
140	996.6	b	212.8	b c
112	1029.5	b	215.6	a b c
56	993.5	b	215.2	a b c
0	927.4	c	207.1	c
LSD ($p < 0.05$)	56.5		10.2	

* Means followed by the same letter are not significantly different.

Both rate of applied potassium fertiliser and the previous fertilisation history had significant effects (Table 23) on the potassium content of biomass in experiment 1, but the treatment interaction was not significant. Potassium concentration in biomass increased with rate of applied potassium fertiliser. Potassium concentration in biomass of 140, 168 and 224 kg K/ha treatments was significantly higher than 0 and 56 kg K/ha treatments. Differences between 112 and 140 kg K/ha treatments or between 56 and 112 kg K/ha treatments were not significant. Potassium concentration in biomass also generally increased in response to levels of potassium in soil from previously applied potassium fertiliser. Potassium concentrations associated with soil which previously received 0-84 kg K/ha were not significantly different. However biomass K in previous 168 and 196 kg K/ha soils were significantly different from each other and all other histories.

Table 23. Effect of previous history of potassium fertilisation and applied rate of potassium fertiliser on potassium concentration in biomass of experiment 1.

	Concentration of potassium in biomass (% DM)					
	Previous K fertiliser history (kg K/ha)					
K rate (kg K/ha)	0	28	84	168	196	Mean for rate
0	0.73	0.69	0.62	0.99	0.98	0.80
56	0.73	0.82	0.92	1.08	1.27	0.96
112	0.93	1.13	0.98	1.22	1.24	1.10
140	1.05	1.20	1.04	1.43	1.54	1.25
168	1.22	1.15	1.18	1.43	1.50	1.29
224	1.32	1.17	1.31	1.39	1.69	1.38
Mean for history	0.99	1.02	1.01	1.26	1.37	
LSD (p<0.05) K rate means						0.16
LSD (p<0.05) history means	0.08					

There was a moderate linear association ($r^2=0.45$) between fresh biomass yield and K% in biomass dry matter (Figure 42a). This graphic shows data clusters associated with increasing rate of K fertiliser along the distribution. Most yields of >1000 g/pot were associated with >1% K in dry matter. The majority of these data were derived from application of ≥ 112 kg K/ha. Re-examination of these data to highlight previous K fertiliser history (Figure 42b) showed all histories were associated with the above yield and K% DM criteria. This is a visual reinforcement of the absence of a significant history effect on yield. However it should be noted that highest yield and K% DM was associated with the combination of the highest K fertiliser rate and the highest K fertiliser history.

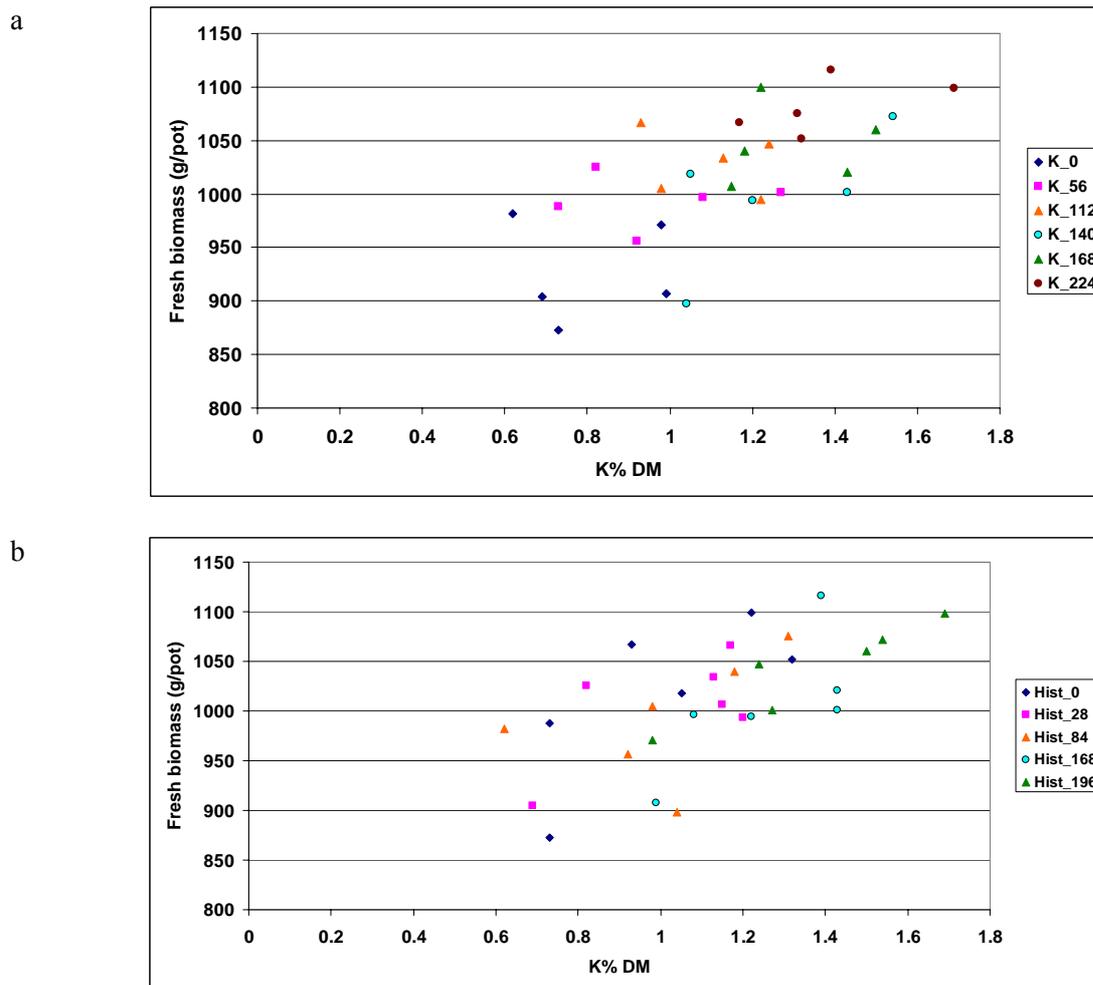


Figure 42. Effect of K% in biomass dry matter on fresh cane yield. (a) highlights K rate data, while (b) highlights previous K fertiliser history.

Both rate of applied potassium fertiliser and the previous fertilisation history had significant effects on levels of exchangeable and reserve potassium (nitric K) in soil in experiment 1 (Tables 24a, b), but the treatment interaction was not significant. Both soil potassium indices generally increased in response to applied fertiliser (Table 24a). The previous rate history had similar effects on exchangeable K, but effects on nitric K were inconsistent (Table 24b).

Table 24. (a) Effect of applied fertiliser on exchangeable and reserve (Nitric K) potassium in soil at the end of experiment 1.
(b) Effect of previous potassium fertiliser history on exchangeable and reserve (Nitric K) potassium in soil at the end of experiment 1.

(a)

Fertiliser rate (kg K/ha)	Exchangeable K (cmol _c /kg)	Significance of means (Exch K)	Nitric K (cmol _c /kg)	Significance of means (Nitric K)
0	0.047	a	1.22	a
56	0.046	a	1.25	a b
112	0.056	b	1.26	a b
140	0.059	b	1.26	b c
168	0.061	b	1.28	b c
224	0.068	c	1.29	c
LSD (p<0.05)	0.006		0.04	

(b)

Historic rate (kg K/ha)	Exchangeable K (cmol _c /kg)	Significance of means (Exch K)	Nitric K (cmol _c /kg)	Significance of means (Nitric K)
0	0.045	a	1.086	a
28	0.054	b	1.344	b
84	0.053	b	1.251	c
168	0.063	c	1.337	B
196	0.066	c	1.288	D
LSD (p<0.05)	0.005		0.023	

Experiment 2: Effect of cane trash on soils with differing history of K inputs

As for experiment 1 there was no significant effect ($p < 0.05$) of previous history of potassium fertilisation or rate of applied trash on dry matter yield at 175 days after planting; there was also no consistent ranking of dry matter yield in relation to these variables. Rate of trash application had no significant effect on fresh biomass yield in this experiment, but there was a significant effect ($p = 0.06$) of previous fertiliser history, with yield generally increasing with rate of previously applied potassium fertiliser, up to 168 kg K/ha (Figure 43). The 28 kg K/ha result was anomalous in this sequence. Yield for 28, 112, 140, 168 and 196 kg K/ha soil histories were all significantly greater than the previously unfertilised control soil.

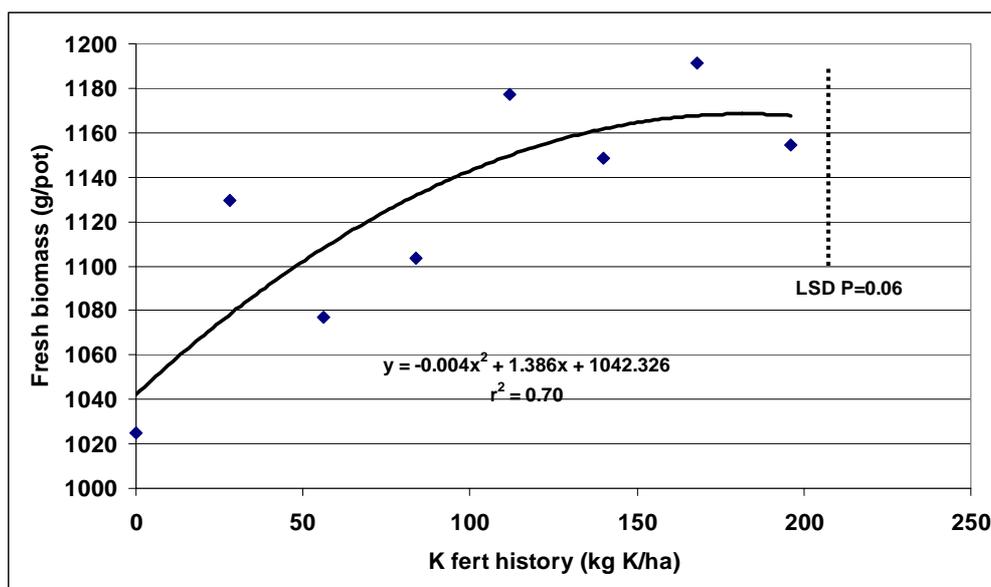


Figure 43. Response of fresh biomass yield to previous history of potassium fertilisation.

Both rate of applied trash and previous fertiliser history had significant effects ($p < 0.05$) on biomass moisture content in experiment 2, wherein the 10 and 20 t trash /ha treatments had a significantly higher water content than did the zero trash treatment (Table 25). While the analysis of variance procedure indicated a significant effect of previous potassium fertiliser history on dry matter content, there was inconsistency in the response. The 0 and 56 kg K/ha previous potassium treatments had significantly lower water contents than did the 84 to 196 kg K/ha histories, but 0 and 28 kg K/ha were also not different from the 84 to 196 kg K histories (Table 25).

Potassium concentration in dry biomass was a significant function ($p < 0.05$) of both rate of applied trash and the previous fertiliser history (Table 26). Potassium concentration increased with rate of application for both variables. Ten and 20 t trash/ha were associated with significantly higher potassium levels in biomass than were the 0 and 5 t/ha rates. There were six homogenous groups of potassium concentrations with respect to previous fertiliser history, with rates greater than 84 kg K/ha being significantly higher than the 0 kg K/ha treatment.

Table 25. Effect of applied rate of trash and previous potassium fertiliser history on moisture content of sugarcane biomass in experiment 2.

Rate of trash (t/ha)	Moisture content (%)	Significance of means (Moist %)	Previous history (kg K/ha)	Moisture content (%)	Significance of means (Moist %)
0	77.6	a	0	77.8	a b
5	78.1	a b	28	78.1	b
10	78.4	b	56	76.9	a
20	78.4	b	84	78.7	b
			112	78.7	b
			140	78.0	b
			168	78.3	b
			196	78.7	b

Table 26. Effect of applied rate of trash and previous potassium fertiliser history on potassium concentration in dry sugarcane biomass in experiment 2.

Rate of trash (t/ha)	K conc. (%DM)	Significance of means (K % DM)	Previous history (kg K/ha)	K conc. (%DM)	Significance of means (K % DM)
0	0.65	a	0	0.61	a
5	0.67	a	28	0.64	a b c
10	0.77	b	56	0.62	a b
20	0.86	b	84	0.72	b c d
			112	0.75	c d e
			140	0.77	d e
			168	0.85	e f
			196	0.93	f

Both exchangeable and nitric acid extractable (nitric K) potassium concentrations in soil responded significantly ($p < 0.05$) to rate of applied trash and the previous fertiliser history (data not shown). The 0 and 5 t trash/ha supported significantly lower exchangeable soil potassium values compared to the 10 and 20 t/ha treatments. Values were similar within the former pair of rates. The 0 and 28 kg K/ha historical treatments had significantly lower exchangeable potassium than did rates > 56 kg K/ha. All rates of trash treatments had significantly higher nitric K than the 0 trash treatment. The response of nitric K to historical treatments was not a consistent function of rate, but 0 kg K/ha had a significantly lower nitric K value than did all treated soil.

Release of potassium from the applied trash is summarised in Table 27.

Table 27. Release of potassium from trash applied to experiment 2, where all trash rates received the same amount of irrigation.

Rate of trash (t/ha)	K conc. (%DM)	Significance of means (K% DM)	% of original K concentration	Significance of means (% orig. K)	kg K/ha released from trash
5	0.19	a	24.3	a	29.7
10	0.32	b	40.7	b	31.8
20	0.42	c	54.0	c	84.3

Experiment 3: Availability of trash-K to sugarcane

There were no significant differences in fresh or dry biomass yield or potassium concentration in biomass ($p = 0.74$ and 0.66 respectively) associated with potassium leached from trash by 10, 25, 40, 62.5 or 115 mm of simulated rainfall. Residual potassium concentration in trash was a significant function of the simulated rainfall treatments, but only at $p = 0.09$ (Table 28). However the water treatments had minimal effect of removing potassium from trash.

Experiments 4 and 5: Assess the fate of K leached from trash

Experiment 4 was conducted with the illitic Chromosol soil from BSES Mackay and low to moderate exchangeable potassium and high levels of nitric K, while experiment 5 was conducted to contrast the latter soil with a Kurosol soil with low levels of both exchangeable and nitric K.

Removal of potassium from the trash mulch was a significant function ($p < 0.05$) of the quantum of applied irrigation (Figure 44) for both experiments; loss of potassium increased significantly with each increment of irrigation up to 150 mm, but there was no significant difference between losses associated with 150 and 200 mm of water (LSD not shown).

Table 28. Effect of simulated rainfall on residual potassium in trash and the release from trash in experiment 3.

Rainfall (mm)	Residual K in trash (% DM)	Significance of means (K % DM)	kg K/ha released from trash
10	0.87	a	0
25	0.79	a b	3
40	0.68	b	15
62.5	0.68	b	15
115	0.62	b	21

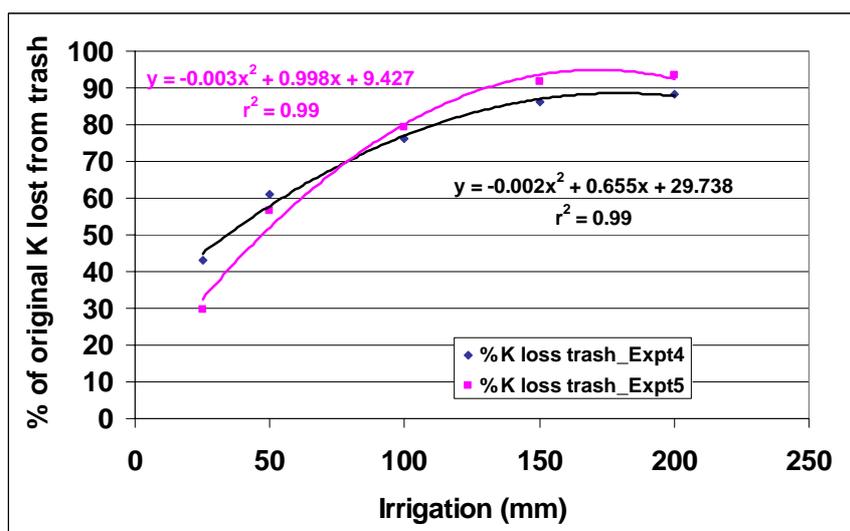


Figure 44. Impact of irrigation regime on the leaching of potassium from a trash blanket in experiments 4 and 5.

Both indices of soil potassium status demonstrated significant responses ($p < 0.05$) to the application of trash and the varying amounts of trash potassium leached by irrigation water, (Table 29) and also for the depth from which the sample was taken in the pot (Table 30). Application of trash resulted in a significant increase in levels of exchangeable potassium in both experiments (Table 29 - Control_start and Control_end are the bare soil treatments). Exchangeable potassium averaged across depths generally increased in response to the quantity of applied irrigation water, but in both experiments 100 mm of water resulted in higher soil potassium than did 200 mm, but the values were not significantly different. Nitric K levels also increased in response to potassium leached from the trash, but 50 mm or more were required for the increase to be significantly greater than the starting value.

Exchangeable and nitric K values point to stratification of potassium particularly in the 0-2 cm zone for both experiments, where potassium levels are significantly higher than for other depth intervals (Table 30).

There was a significant interaction ($p < 0.05$) between water treatment and depth interval for both indices of soil potassium status in experiments 4 and 5. This interaction is a reflection of the above statement about potassium stratification, in that shallow layers showed a greater increase in potassium status than did deeper layers, and there were also significant differences in potassium levels within layers due to differential delivery of trash potassium in water treatments (Table 31).

Table 29. Exchangeable and nitric K data for experiments 4 and 5 in relation to water management treatment.

Treatment	Exchangeable K (cmol _e /kg)				Nitric K (cmol _e /kg)			
	Experiment 4		Experiment 5		Experiment 4		Experiment 5	
	Exch. K	Signif. of means	Exch. K	Signif. of means	Nitric K	Signif. of means	Nitric K	Signif. of means
100 mm	0.31	a	0.18	a	1.25	a	0.32	a
200 mm	0.29	a b	0.17	a b	1.24	a b	0.29	a b
150 mm	0.28	b	0.16	a b	1.20	b c	0.28	b
50 mm	0.27	b	0.15	b c	1.19	c d	0.27	b
25 mm	0.24	c	0.13	c	1.16	d e	0.23	c
Control_start	0.21	d	0.10	d	1.13	e f	0.20	c d
Control_end	0.19	d	0.09	d	1.10	f	0.18	d

Table 30. Exchangeable and nitric K data for experiments 4 and 5 in relation to depth of sampling in the pot.

Treatment	Exchangeable K (cmol _e /kg)				Nitric K (cmol _e /kg)			
	Experiment 4		Experiment 5		Experiment 4		Experiment 5	
	Exch. K	Signif. of means	Exch. K	Signif. of means	Nitric K	Signif. of means	Nitric K	Signif. of means
0-2 cm	0.33	a	0.17	a	1.25	a	0.30	a
2-5 cm	0.24	b	0.14	b	1.16	b	0.25	b
5-10 cm	0.22	c	0.12	c	1.15	b	0.23	b c
10-25 cm	0.23	c	0.12	c	1.16	b	0.22	c

Table 31. The interaction between water treatments and depth interval for the effect of potassium from trash on exchangeable K in soils for experiments 4 and 5.

Treatment	Exchangeable K (cmol _e /kg)							
	Experiment 4				Experiment 5			
	0-2 cm	2-5 cm	5-10 cm	10-25 cm	0-2 cm	2-5 cm	5-10 cm	10-25 cm
Control_start	0.20	0.20	0.20	0.23	0.11	0.11	0.09	0.10
25 mm	0.33	0.23	0.21	0.21	0.16	0.12	0.12	0.11
50 mm	0.41	0.27	0.24	0.22	0.17	0.14	0.14	0.14
100 mm	0.47	0.28	0.25	0.25	0.27	0.19	0.15	0.14
150 mm	0.38	0.25	0.22	0.22	0.23	0.15	0.13	0.12
200 mm	0.39	0.28	0.24	0.23	0.23	0.18	0.14	0.13
Control_end	0.14	0.18	0.22	0.23	0.08	0.08	0.09	0.10
LSD (p<0.05)	0.03				0.04			

Soil potassium data averaged across depth intervals (Table 29) did not indicate any significant difference in exchangeable or nitric K values between the control_start treatment (25 mm of pre-leaching and before application of trash) and the control_end treatment (bare soil after application of 25 mm pre-leaching and 200 mm of “irrigation” water). These data could be interpreted to imply there was no leaching of soil potassium. However reference to the more detailed depth data in Table 31 shows there was significant leaching of exchangeable K from the 0-2 cm zone and a near significant loss from the 2-5 cm zone in the bare soil treatments by an additional 200 mm of water in experiment 4; nitric K data showed a similar response, but effects were significant for both 0-2 and 2-5 cm intervals. A similar but non-significant effect was noted for experiment 5. Leachate data from the “Control” treatments also demonstrates leaching of potassium from bare soil.

The bare soil “control” treatments at the start and end of the experiment were included to separate effects of potassium leached from soil and additional potassium leached from applied trash. During the interpretation of the data it was realised that there should have been a bare soil “control” associated with each destructive recovery of trash when the target irrigations were achieved. This deficiency in experimental design was addressed through a potassium balance on the leachate for each treatment (corrected for potassium in the applied water). Potassium in bare soil leachate was then expressed as a function of applied water (Figures 45a, b) to derive potassium values for bare soil leachate for each water treatment. These were then deducted from the potassium due to soil and trash sources to arrive at estimated loss of trash potassium from the 0-25 cm interval (plough layer equivalent) in Table 32.

Data in Figure 45 show that illitic Chromosol soil (Figure 45a) leached at larger quantum of potassium from bare soil than did the lower fertility Kurosol (Figure 45b), presumably due to the higher concentration of potassium in the soil solution of the illitic soil after mineralisation of potassium from a large reserve of non-exchangeable potassium. Rates of loss of soil and soil + trash potassium were similar for the illitic soil (0.181 and 0.173 kg K / mm of water respectively), whereas loss rate for the soil + trash potassium in the Kurosol was almost twice that of the bare soil alone (0.066 and 0.036 kg K /mm water respectively). Data from Figure 45 were used to estimate the quantity of trash potassium that was leached (Table 32).

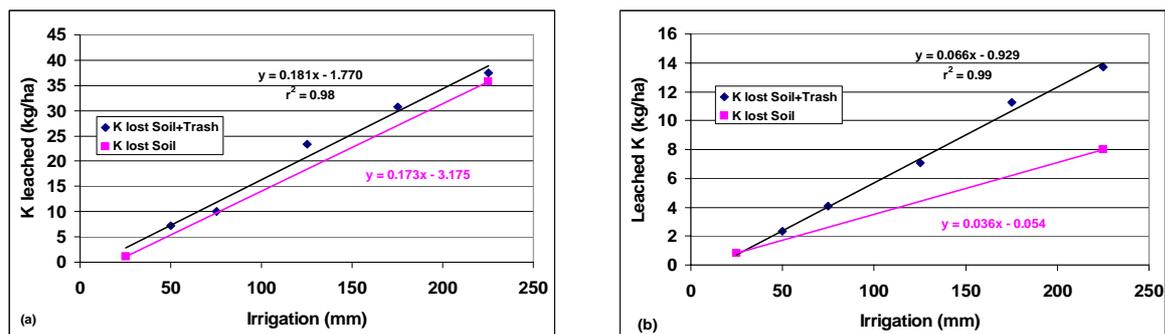


Figure 45. Relationships between applied water and potassium lost to leachate from soil and soil + trash sources for the Chromosol - experiment 4 (a) and the Kurosol experiment 5 (b).

Table 32. Estimates of potassium sources from the trash blanket that has been leached below 25 cm in experiments 4 and 5.

Treatment	Potassium from trash blanket leached from pots (kg K/ha)	
	Experiment 4 Chromosol	Experiment 5 Kurosol
25 mm	1.8	0.6
50 mm	2.0	1.4
100 mm	2.4	2.9
150 mm	2.8	4.4
200 mm	3.2	5.9

Discussion

Field experiment

In the field K experiment, a yield plateau was reached with increasing K fertiliser application, with maximum yields occurring at application rates of 120 kg ha^{-1} in 1999 (Figure 37b), and 40 kg ha^{-1} in 2000 and 2001 (Figures 37c and d). The lower K rates at maximum yield in the 2000 and 2001 crops reflect the markedly lower yields in these years compared with the 1999 crop (Figure 37) due to the orange rust infestations in the 2000 crop and the dry conditions in the 2001 crop. Despite a yield plateau being achieved in all three crops, there was no evidence that maximum yields were attained at lower rates of K application in the trash retained treatments – the mean yield difference between the trash retained and removed treatments was similar at all K application rates. This result would not have been expected if the K in the trash blanket ($40\text{-}100 \text{ kg ha}^{-1}$) was available to the crop in the short-term. The presence of trash will affect more than soil fertility, however. Trash suppresses evaporation of water from the soil so that the trash retained plots of the 1999 and 2000 crops may have benefited from both the nutritional and water conservation properties of the trash blanket.

This was not the case for the 2001 crop however, as trash retention was achieved through trash incorporation prior to planting and so soil water balances of the two trash treatments would have been more similar than in previous crops. In the 2001 crop there was still no evidence that maximum yields were reached at lower K application rates in the trash retained

treatments, although there was still a significant *overall* yield benefit of trash retention (Figure 37d). The reasons for this yield benefit are unclear, as discussed further below.

Despite the lack of clear K-benefit in the yield results, there is obvious evidence that K from retained trash entered the soil-plant system. K concentrations in cane of the crops harvested in 1999 and 2000 increased significantly ($P < 0.05$) with both increasing K applications and trash retention (Figures 41a and b). The result was similar for trash removed from the plants prior to harvest in 2000 (Figure 40a). K concentrations of the harvested cane were significantly, linearly related to K applications rates. The difference in cane K concentration between the trash retained and removed treatments can be compared with the relationship between K concentrations and applications rates to provide insight into the amount of fertiliser K that would have had to be applied to obtain the equivalent increase in cane K concentration. For both crops this amount was approximately 42 kg ha^{-1} . This amount compares with a site mean amount of K in the trash of 46 and 103 kg ha^{-1} at the start of the 1999 and 2000 crops, respectively.

Interestingly for the Q138 plant crop, for which trash was incorporated into the soil, there was no significant difference between cane K concentrations of the two trash treatments (Figure 41c). The result was similar for trash removed from the plants prior to harvest in 2001 (Figure 40b). These results suggest that the incorporation and/or plough out-planing operations may have redistributed incorporated trash, or the K released from it, across the experimental plots. It also suggests that the response of cane yield to trash retention (Figure 37d) may not have been related to K nutrition.

While the results of this experiment have shown that (1) cane (and sugar) yields respond significantly to the retention of trash and (2) K concentrations in sugarcane plants increase significantly where trash is retained as a blanket, the cause of the yield response can not be directly attributed to K in the trash blanket as cane yields were not significantly increased by the retention of trash at sub-optimal rates of K fertiliser application. The yield response may have been due to the soil water conservation benefits of trash blanketing, or some other factor. It is possible that the conditions during the experiment contributed to the lack of causal link between trash blanketing and yield response to K fertiliser. The crop harvested in 1999 had been stood over, lodged badly and there was considerable variation in yield between replicate plots. Yields in the following crops were low and variable due to orange rust and dry conditions, and the maximum yields were attained at the lowest rate of applied K (40 kg ha^{-1}). Thus the experimental design could not discriminate a trash response at lower rates of K fertiliser. However, given the clear response in plant K concentrations to trash blanketing, it is likely that K fertiliser application rates can be reduced in trash blanketed crops compared to the appropriate rate for burnt crops.

K release from trash blankets and stratification in soil

The fresh weight criterion for discussion of yield data is relevant because of its commercial significance to the sugar industry. The increase in moisture content associated with higher potassium fertiliser rates may be a reflection of the role of this element in the osmotic adjustment of sugarcane. Similar effects were observed in experiments 1 and 2.

Experiments 1 and 2 demonstrated that previous history of potassium fertilisation has had a significant effect on prevailing levels of exchangeable soil potassium after the harvest of the pot experiment, an effect previously reported by Chapman (1980) for the site used to supply soil for these experiments. History of fertilisation however was not reflected in the yield response for experiment 1, where applied fertiliser gave significant responses in fresh cane yield. The rate of applied trash did not have an effect on yield in experiment 2, whereas the

previous history of potassium fertiliser application did influence yield. Applied potassium fertiliser (experiment 1) and applied trash (experiment 2) were both associated with a rate response in the potassium concentration in biomass. Therefore because of effect on soil and plant potassium levels we can be confident that the potassium leached from the trash has direct nutrient value, as with the field experiment discussed above.

Why then was a potassium fertiliser response effect noted for experiment 1, yet no trash rate effect in experiment 2? This mainly due to variability of data in experiment 2 associated with waterlogging in several pots, but the issue is further explained in part by examination of potassium concentrations in biomass (Table 33) and soil exchangeable potassium values (Table 34) in relation to applied fertiliser and trash across previous fertiliser histories that were common to experiments 1 and 2. Assumption of a linear effect of applied potassium allows data for the 10 and 20 t trash/ha to be averaged to apply 59 kg of trash potassium for comparison with 56 kg K of fertiliser potassium. Fertiliser and trash potassium had similar effects on biomass potassium within the 0 and 28 kg K/ha histories, but fertiliser potassium caused slightly higher biomass potassium values for 84, 168 and 196 kg k/ha histories, but did not result in higher yield. Exchangeable potassium values were very similar for relevant comparisons across experiments. Experiment 1 data showed 0 kg/ha fertiliser K produced significantly lower yield than the other rates of fertiliser, however there was no significant difference in yield between 56 and 168 kg K/ha. The maximum rate of trash K applied in experiment 2 was 72 kg K/ha. Comparison of the two data sets for relative yield response to potassium in dry biomass (Figure 46) associated with 56 or 59 kg K/ha from fertiliser or trash K respectively, shows that $Y_{100\%}$ was obtained at similar potassium concentrations (0.82 and 0.84%). In fact these similar rates of potassium from fertiliser and trash produced $Y_{100\%}$ and $Y_{97\%}$ for experiments 1 and 2 respectively with the latter K% DM values in the 28 kg K/ha prior history treatment. Therefore we can have every confidence that potassium released from the trash has equivalent value to fertiliser potassium for nutritional purposes.

The significant difference in concentration of residual potassium in the rates of trash treatments in experiment 2 (Table 27) for a constant application of water implies that more water will be required to release most of the potassium associated with residues of high yielding crops.

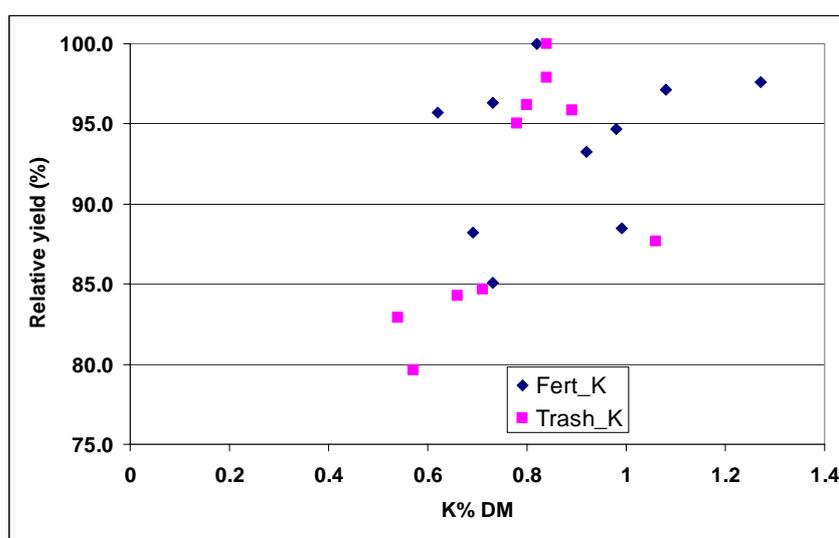


Figure 46. Comparison of the response of relative yield of fresh biomass to potassium concentration in dry biomass for 0 and 56 kg fertiliser K/ha in experiment 1 and 0 and 59 kg trash K/ha in experiment 2, for soils with five previous potassium fertiliser histories that were common to both experiments.

Table 33. Comparison of impact of equivalent quantities of potassium from fertiliser and trash sources on potassium concentration in biomass.

		Concentration of potassium (% DM)				
		Previous K fertiliser history (kg K/ha)				
Expt 1	Fertiliser K (kg K/ha)	0	28	84	168	196
	0	0.73	0.69	0.62	0.99	0.98
	56	0.73	0.82	0.92	1.08	1.27
Expt 2						
Trash rate (t/ha)	Trash K (kg K/ha)					
0	0	0.54	0.57	0.66	0.80	0.84
10	46	0.61	0.65	0.77	0.84	1.10
<i>15*</i>	<i>59</i>	<i>0.71</i>	<i>0.84</i>	<i>0.78</i>	<i>0.89</i>	<i>1.06</i>
20	72	0.80	1.03	0.79	0.93	1.02

* interpolated value

Table 34. Comparison of impact of equivalent quantities of potassium from fertiliser and trash sources on exchangeable potassium in soil.

		Concentration of potassium (% DM)				
		Previous K fertiliser history (kg K/ha)				
Expt 1	Fertiliser K (kg K/ha)	0	28	84	168	196
	0	0.035	0.045	0.044	0.054	0.057
	56	0.038	0.045	0.042	0.054	0.052
Expt 2						
Trash rate (t/ha)	Trash K (kg K/ha)					
0	0	0.038	0.040	0.043	0.051	0.053
10	46	0.040	0.047	0.047	0.058	0.066
<i>15*</i>	<i>59</i>	<i>0.042</i>	<i>0.048</i>	<i>0.047</i>	<i>0.058</i>	<i>0.064</i>
20	72	0.043	0.048	0.046	0.057	0.061

* interpolated values

Exchangeable K and Nitric K levels in the soil at the end of experiments 1 and 2 generally were substantially lower than the initial values (Table 17). This may be a reflection of the greater exploitation of soil potassium in a restricted root zone in pots when compared to the field. The observation of the combined effect of rate of applied potassium fertiliser and the previous fertiliser history on yield and K% DM was a consequence of the combined impact of these treatments on levels of exchangeable K in soil (Tables 24a, b).

Experiment 3 did not produce any data that contributed to achievement of project objectives. Data in Table 28 show the equivalent of only 21 kg/ha K was leached from the trash by the 115 mm simulated rainfall treatment. Results from experiments 1 and 2 suggest this would have been insufficient to result in a biomass yield response. No data based explanation is available for the low release of potassium (only 25% of the original amount) by the 115 mm treatment, given earlier experience in this project where 50 mm of irrigation leached 90% of

trash potassium and in experiments 4 and 5 where 115 mm of water would have released approximately 80% of the potassium originally contained in trash. Our observations indicate that the size and duration of wetting events may be relevant to the above issues. The high removal of potassium by 50 mm of irrigation and in experiments 4 and 5 was associated with either a single significant wetting event or ponding of water on the soil surface. Cumulative water volume alone is therefore not necessarily a good index of potassium release.

Experiments 4 and 5 provided clear evidence of stratification of the potassium from crop residues in the 0-2 and to a lesser extent in the 2-5 cm zone of both soils in the absence of crops. There was also clear evidence of leaching of potassium that had been mineralised from soil reserves. The authors are not aware of any previous quantification of potassium leaching from soils within the Australian sugar industry. The other point of interest associated with this finding was the insensitivity of bulk soil analysis compared to collection of leachate in detecting the loss of potassium from the bare soil treatments.

Leachate and soil data from experiments 4 and 5 suggest the illitic Brown Chromosol soil may be better placed to support a higher equilibrium relationship between soil solution potassium and the gradual release of potassium from trash, whereas the lower cation exchange capacity Kurosol can support a lower equilibrium concentration thus trash potassium from trash was more prone to leaching than was the soil potassium. This is demonstrated in Table 32 where the Kurosol supports a higher loss of trash potassium than does the Brown Chromosol at the two higher water rates. However in a “non-cropped” system the Brown Chromosol lost more potassium than did the Kurosol. Leached values for trash potassium were less than 10% of the potential 79 kg K /ha in the trash blanket. These experiments were conducted in the absence of plants, so there is every chance the potassium would be accessed by the crop and not be a true leaching fraction.

Conclusion

K from trash is plant-available, so fertiliser and trash K should have similar value as nutrients. However, there was not a significant yield response to K from trash in either pot or field experiments, possibly due to variability in the results. Thus it is not clear that recommendations of K management in trash-blanketed crops should differ from those for burnt crops.

12. SIMULATED IMPACT OF TRASH MANAGEMENT ON SUGARCANE PRODUCTION AND NITROGEN MANAGEMENT

Introduction

As described above, many field experiments have been conducted comparing GCTB and burnt systems. In these experiments, increases in soil organic matter and total N in trash blanketed treatments are small, confined to the surface soil, and poorly related to the age of the experiments. However, microbial activity is clearly stimulated by trash blanketing, even after only short periods of trash retention. Ng Kee Kwong *et al.* (1987) and Basanta *et al.* (2003) showed that crop uptake of N from a trash blanket in the season following deposition of the trash was negligible and the N was mainly immobilised in the soil organic matter.

Despite the insights gained in these studies, it is still uncertain how N management recommendations for GCTB systems should differ from those for burnt systems. If N returned to the soil through trash blanketing becomes available to the crop, there may be yield increases in situations when the crop would otherwise be N-limited. For example, yields in systems where trash is retained might be higher than in burnt systems at lower rates of N fertiliser. Or, the N rate at which sugarcane yields cease to respond positively to applied N (i.e. the N rate at which yields 'plateau') might be lower when trash is retained. However, the trash management experiments conducted to date have been undertaken at a single rate of applied N fertiliser, and so comparative experimental N response data for GCTB and burnt systems do not exist.

The complexity of the N cycle and the long time scales involved in soil organic matter cycling make simulation-based approaches to this N management problem attractive. During this project, considerable progress was made in our ability to simulate sugarcane production systems, including their response to variations in N and trash management. In this section of the project, we take advantage of these advances and use a simulation approach to gain insights into the long-term fate of N contained in trash and identify the N fertiliser management implications of trash retention.

Methodology

The general approach of this study was to simulate sugarcane yield and environmental N losses at different rates of N fertiliser applied to three different trash management systems over a wide range of climates and soils types.

Simulation details

The three systems simulated were 1) trash burnt at harvest, 2) trash retained at harvest and 3) trash retained at harvest with cane yields limited to equal yields from burnt system.

Simulations were conducted for a combination three different soil types and five different climates, giving 15 soil type–climate combinations. The soil types were (according to the Australian Soil Classification) a Red Kandosol, a Brown Chromosol and a Brown Dermosol (Table 35). The soils were chosen to represent a range of soils found in the Australian sugar industry (although not the entire range) and, importantly, because data were available for the derivation of model parameters (described below).

Table 35. Properties (0-0.2 m) of the three soils used in the simulations.

	Red Kandosol	Brown Chromosol	Brown Dermosol
Details of previous study to provide soil parameters	Thorburn <i>et al.</i> (2003b)	Mackay (Table 11)	Tully (Table 11)
Great soil group	Red earth	Non-calcic brown soil	Chocolate soil
Soil texture	loam/sandy loam	loam/clay loam	silty clay
Organic C (%)	0.8	1.0	1.2
C : N ratio	12.3	18.0	10.3
Bulk density (kg/m ³)	1.50	1.51	1.32
Drained Upper Limit* (%)	28.4	30.0	36.5
Lower Limit* (%)	12.8	17.0	22.5

*Soil water parameters used in the SoilWat model (Probert *et al.*, 1998), analogous to field capacity and wilting point.

Climate data were obtained from the Australian Bureau of Meteorology Silo database for Abergowrie (upper Herbert), Burdekin, Bundaberg, Mackay and Tully. These locations were chosen to provide a range in climates, and all, except Bundaberg, were the sites of previous investigations of trash blanketing (e.g., Table 11). For the Burdekin and Bundaberg climates, irrigation was applied in the simulations. Irrigation was applied when the soil dried to a pre-determined extent. Total irrigation applications within a crop were capped at 800 mm for the Burdekin climate and 400 mm for the Bundaberg climate. Simulations for the other three climates had no irrigation.

N fertiliser application rates were varied from 0 to 300 kg/ha (in 30 kg/ha increments) on ratoon crops, with plant crops receiving 75 % of that applied to the ratoon crops (as is recommended N management). N was applied as urea at 75 mm soil depth, 8 weeks after planting/harvest.

Trash burning was simulated through removal of 95 % of trash at harvest, equivalent to trash burnt both pre- and post-harvest (Mitchell *et al.*, 2000).

For trash management Systems 1 (trash burnt) and 2 (trash retained), potential sugarcane yields were predicted. That is, there was no allowance made in the simulations for factors such as pests and diseases, lodging, water logging, stool damage during harvest, etc., which will limit yields in the field. This resulted in yields in the trash retained system being greater than those in the burnt system in most circumstances. Trash blanketing does not always result in greater yields, particularly in soils with poor drainage (Wood, 1991) and cool areas (Kingston *et al.*, 1998). N exported from the field in harvested cane is a large term in the N balance of a sugarcane cropping system, so N recommendations derived from simulations of trash blanketed systems where trash blanketing increases yield will not necessarily be applicable where trash blanketing does not increase yield. Thus for trash management System 3, cane yields were limited to be similar to those of the burnt system by reducing the radiation use efficiency (RUE) parameter in APSIM-Sugar by a fixed proportion throughout the simulations. The reduction was determined by approximately matching cane yields in trash management System 3 to those in System 1 at the highest rate of applied fertiliser N (300 kg/ha). A specific RUE reduction was determined for each climate-soil type

combination and both plant and ratoon crops (Table 36). This reduction was then used to reduce yields for all rates of fertiliser N when simulating trash management System 3.

Table 36. Average reduction (%) in RUE used to simulate yields for trash management System 3 for simulated plant crops (P) and ratoon crops (R).

Soil type	Climate data									
	Abergowrie		Burdekin		Bundaberg		Mackay		Tully	
	P	R	P	R	P	R	P	R	P	R
Red Kandosol	3	20	12	13	12	25	0	17	2	5
Brown Chromoso 1	3	22	10	12	12	25	0	17	2	5
Brown Dermosol	3	22	8	14	14	25	0	17	2	3

The simulations were conducted over 100 years. For the 1st 25 years, the trash and N fertiliser management system was the same in all simulations – trash burnt with 160 kg/ha of N fertiliser applied to ratoon crops and 75 % of this applied to plant crops. This was done for two reasons: (1) to allow the simulated plant-soil system to come to some equilibrium and not be overly affected by the impacts of the initial values chosen for the model parameters, and (2) to represent the dominant trash management used in the industry prior to the adoption of GCTB. At the end of 25 years, the different trash and N fertiliser management systems were introduced to the simulations.

Multiple simulations were conducted and the results averaged to overcome bias possibly caused by temporal patterns in climate or the impacts of different cropping cycle specifics (e.g., planting time, fallow length, etc.). Two cropping cycles were simulated. The first consisted of an autumn plant crop (15 months long) followed by four 13 month ratoon crops and a short fallow. The second consisted of an early spring plant crop followed by four 13 month ratoon crops and a longer fallow. Simulations were started in two years, 1901 and 1904.

Model configuration

Simulations were undertaken with the APSIM cropping systems simulator. The model was configured to consist of modules for soil N and C dynamics (APSIM-SoilN; Probert *et al.*, 1998), soil water dynamics (APSIM-SoilWat; Probert *et al.*, 1998), sugarcane residue decomposition (APSIM-Residue; using the sugarcane-specific developments detailed in other sections of this report), and sugarcane growth (APSIM-Sugarcane; Keating *et al.*, 1999). The modules are one-dimensional, use a daily time-step and are driven by climatic data. The dynamics of water, N, C and roots are simulated in soil layers, with water (and associated nitrate) moving between layers where gradients exist. N mineralisation, N immobilisation and nitrification are explicitly described in each layer, as are the N losses from denitrification and leaching. Soil moisture and temperature affect all soil N cycling processes. The soil water module is a “cascading bucket” water balance model. The presence of plant residues on the soil surface affects runoff (and hence infiltration) and evaporation. The sugarcane module uses intercepted radiation to produce assimilates, which are partitioned into leaf, cabbage (defined as the immature top of the stalk plus green leaf sheaths), structural stalk, roots and sugar. These processes are responsive to radiation and temperature, as well as water and N supply. Farming operations (such as fertilisation, planting, incorporation of crop residues

through cultivation, or burning of crop residues) were specified through the APSIM-Manager module.

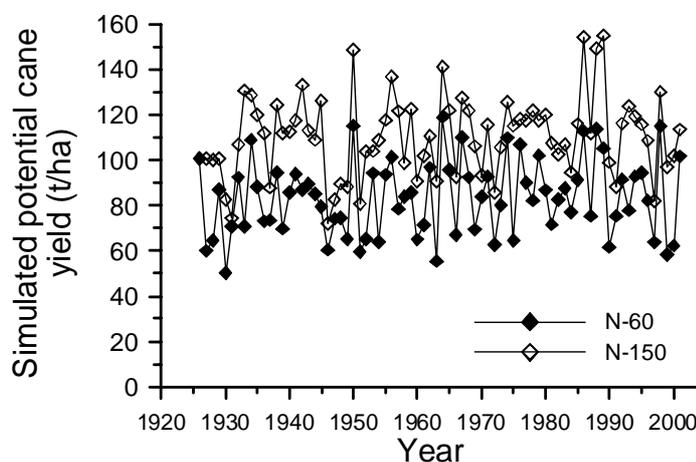
Model parameterisation

Model parameters were generally based on measured soil data. For the Brown Chromosol and Brown Dermosol, these data came from the studies of the impact of trash blanketing on soil N undertaken within this project (as is detailed in Table 35). For the Red Kandosol, data came from a N fertiliser management study (Thorburn *et al.*, 2003b). Values of bulk density, soil C:N ratio and the initial soil organic C (Table 35) were set equal to values measured in the soils. Parameters for the soil water model, drained upper limit and lower limit (Table 35), were derived from measurements of water retention at -33 kPa and -1.5 MPa and, where available, measurements of water content in the field. The layer 'structure' of the soil models was the same for all three soils: the soil was divided into seven soil layers, with a total depth of 1.5 m.

Results

In all simulations, the potential sugarcane yields varied markedly between each crop (an example is shown in Figure 47) as expected, in response to climatic differences over the different growing seasons. This variability did not overshadow the consistent impacts simulated for different trash and N fertiliser managements systems. Thus, for simplicity the simulated average N responses are presented (for example, sugarcane yield shown in Figures 48, 49 and 50).

Figure 47. Example of simulated cane yield each year for two different N fertiliser application rates in a trash blanketed system at Mackay on a Brown Chromosol soil.



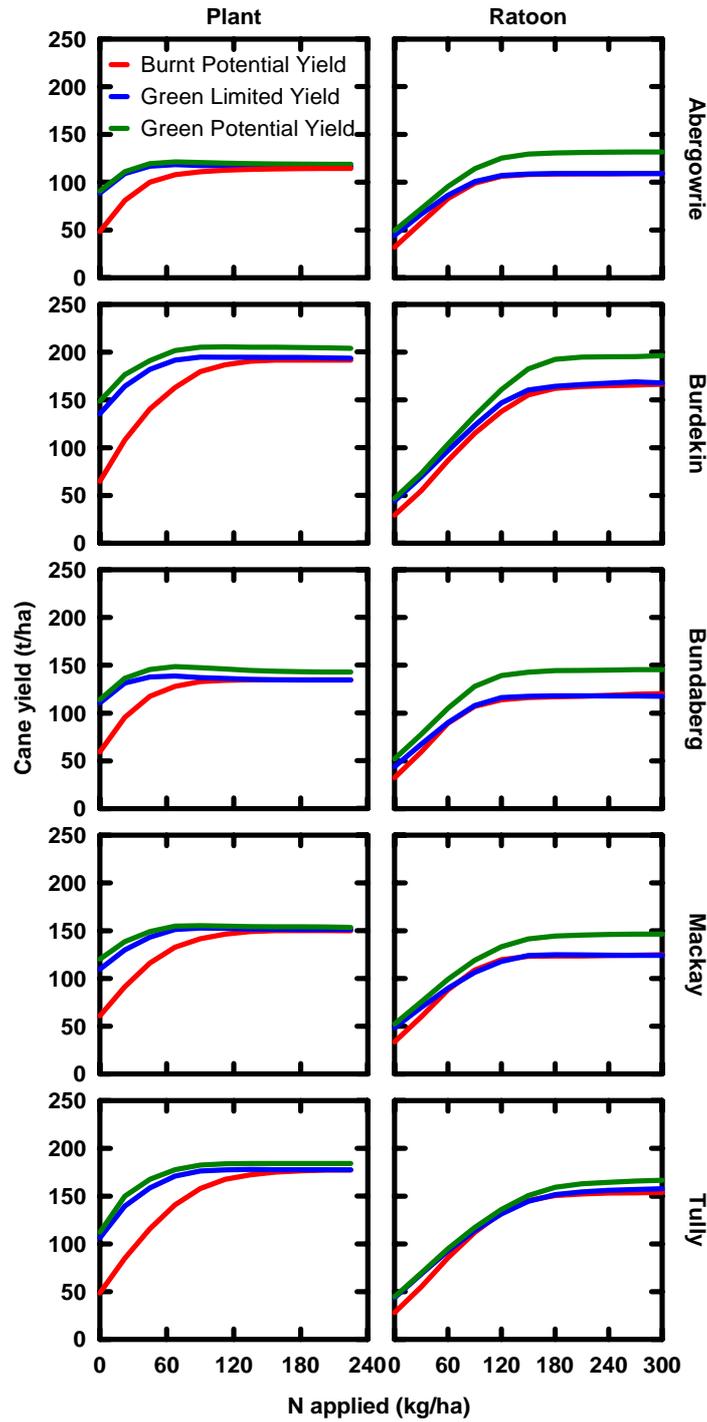


Figure 48. Simulated long-term average sugarcane yield in response to applied N fertiliser for different trash management systems for a Red Kandosol soil.

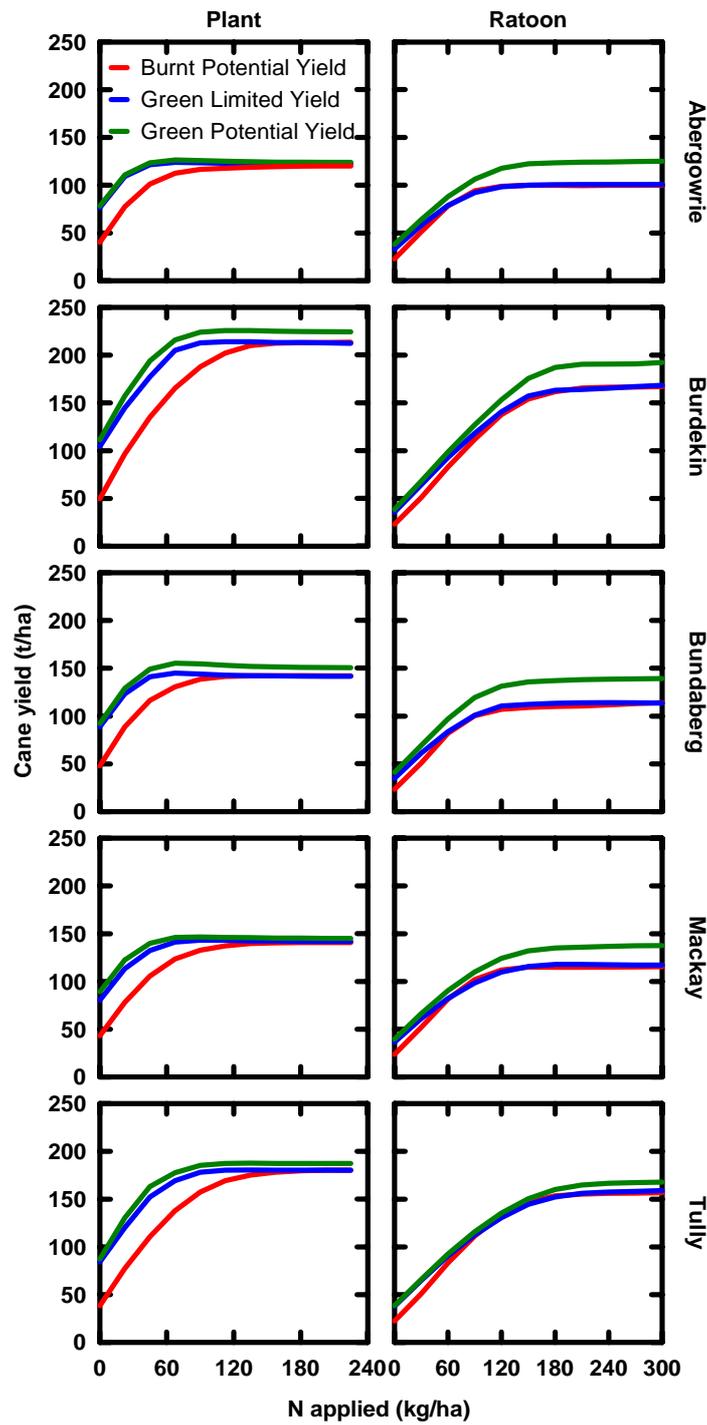


Figure 49. Simulated long-term average sugarcane yield in response to applied N fertiliser for different trash management systems for a Brown Chromosol soil.

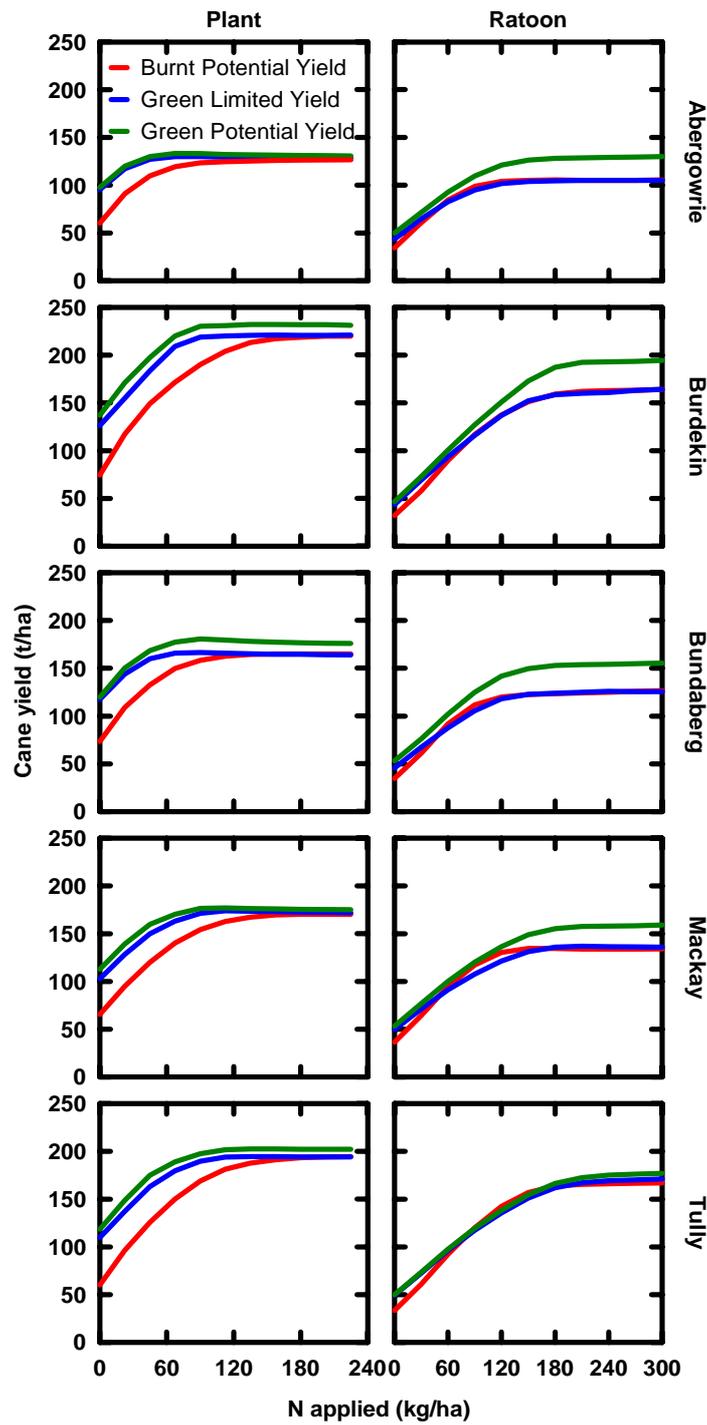


Figure 50. Simulated long-term average sugarcane yield in response to applied N fertiliser for different trash management systems for a Brown Dermosol soil.

Yield response to trash retention

Sugarcane yields (at high rates of N) are simulated to be highest for the Burdekin climate and lowest for the Abergowrie climate, as expected from the radiation and rainfall (and irrigation in the Burdekin) at these locations.

Potential yields were less affected by trash management in the plant crops because of the small amounts of trash on the soil at the time crops were planted in the simulations. Where potential yields are higher with trash retention in the plant crops, at Bundaberg and Burdekin, the increase was again due to moisture conservation in these dry environments. This implies that the amount of irrigation applied in the simulations was insufficient to meet the crops' needs with these two climates.

Generally, *potential* sugarcane yields simulated in ratoon crops for the trash retained system were greater than the burnt systems, the exception being Tully where yield increases were simulated to be small (Figures 48, 49 and 50). The magnitude of the simulated yield increases with trash retained generally agrees with **long-term** experimental results on soils with unrestricted drainage (Wood, 1991; Chapman *et al.*, 2001). The increases in yield are due to lower water stress in the simulations compared with the burnt system. Thus the Tully climate with the highest rainfall (Table 11) had the smallest simulated yield benefit from trash retention.

As described in the Methods section, yields in the Green Limited Yield system were constrained in the simulations to be similar to those of the Burnt Potential system.

N response as affected by trash retention

As expected, sugarcane yields were simulated to increase with increasing amounts of applied N until a yield plateau was reached, and no further response to N occurred (Figures 48, 49 and 50).

Ratoon crops

The N rates at which ratoon crop yields were simulated to plateau in the burnt system depends on the yield. With the Bundaberg and Abergowrie climates, average maximum yields in the burnt system were 100-130 t/ha and these occurred at N rates of 120 kg/ha and above (Figures 48, 49 and 50). Conversely with the Tully and Burdekin climates, average maximum yields were greater than 150 t/ha and these occurred at N rates of 180-200 kg/ha (and above). The yield plateau in the burnt system was reached at the lowest N rates for the Red Kandosol (Figure 48) and highest N rates for the Brown Dermosol (Figure 50). These N rates at which simulated yields plateaued in the burnt system are generally in accord with previous experience with burnt crops in the Australian Sugar industry (Chapman, 1994; Calcino *et al.*, 2000). An exception to this is the Tully climate, where simulated yields and the N rates at which they plateau are much higher than general experience. The simulations do not include processes such as water logging and lodging that limit yields in that district.

For *potential* yields with trash retained, the N rates at which yields plateaued were higher than in the burnt system if ratoon crop yields in the trash retained system were higher than in the burnt systems (Figures 48, 49 and 50). This occurred despite the N recycled in the trash blanket.

With trash retention, when yields were *limited* to be similar to those of the burnt system the N response curve was very close to that for the burnt system (Figures 48, 49 and 50). The main

difference was higher (10-20 %) yields in the Green Limited system than the burnt system at low (e.g. < 60 kg/ha) rates of N.

Plant crops

For the burnt system, the trends in the plant crop N response curves are similar to those in the ratoons crops. The yield plateau was reached at higher N rates where simulated average yields were greater and, in the Brown Dermosol (Figures 48, 49 and 50). Again, these N rates at the yield plateau are in accord with previous experience with burnt crops in the Australian Sugar industry (Chapman, 1994; Calcino *et al.*, 2000).

Unlike the simulations of ratoon crops, the N response curves of plant crops differ considerably in response to trash retention. When trash is retained, the N rate at which simulated yields plateau is markedly lower, i.e. approximately half, than in the burnt system (Figures 48, 49 and 50). Also, at N rates below this, yields are simulated to be higher when trash is retained. These trends occurred in all soil type-climate combinations. Having the yield plateau reached at lower N rates is consistent with the N recycled in the trash retained system 'replacing' N from fertiliser. This effect was only seen in the simulations of plant crops for two reasons. Firstly, the time available for N mineralisation in the plant crops was approximately double that in the ratoon crops because the plant crops would have also been affected by N mineralised during the preceding fallow. Secondly, there were trash blankets in the ratoon crops but not plant crops and decomposing trash immobilises soil mineral N (Ng Kee Kwong *et al.*, 1987; Basanta *et al.*, 2003). Thus these simulations suggest that the impact of trash retention on N fertiliser management will only be seen in plant crops.

Environmental losses of N

As has been simulated previously (Keating *et al.*, 1997), losses of N to the environment by both leaching and denitrification (Figures 51, 52 and 53) increased with increasing rates of applied N fertiliser in all soil type-climate combinations. The losses were markedly greater at N rates greater than those at which yields plateaued. N losses in Plant crops were simulated to be higher than in ratoon crops (Figures 51, 52 and 53).

Total N losses were affected by both soil type and climate, tending to be highest in the Red Kandosol (Figure 51) and lowest for the Burdekin climate (Figure 51, 52 and 53). In the Red Kandosol (Figure 51), the high total losses were mainly due to higher rates of N leaching than in the other soils (Figure 52 and 53). The impact of soil type is more due to soil water characteristics than organic matter. The Red Kandosol has the lowest organic matter (Table 35) but is the most permeable soil of the three soils, with these differences reflected in the SoilWat model parameters. The impact of climate in the simulations is, in the first instance, due to differences in sugarcane yields – climates resulting in higher yields (e.g., the Burdekin) have lower N losses. However, it is also affected by rainfall, with the Tully climate simulated to have higher losses relative to its yield.

In all soil type-climate combinations, N losses are simulated to be higher in the trash retained systems, both potential and limited yields, than the burnt system (Figures 51, 52 and 53). This is so in both plant and ratoon crops. With trash retention, N losses are generally higher with limited yields than potential yields. This is particularly so for denitrification in some of the trash-limited yield simulations. These higher losses in the trash retained systems indicate that not all of the additional N recycled in trash blankets is taken up by the crop.

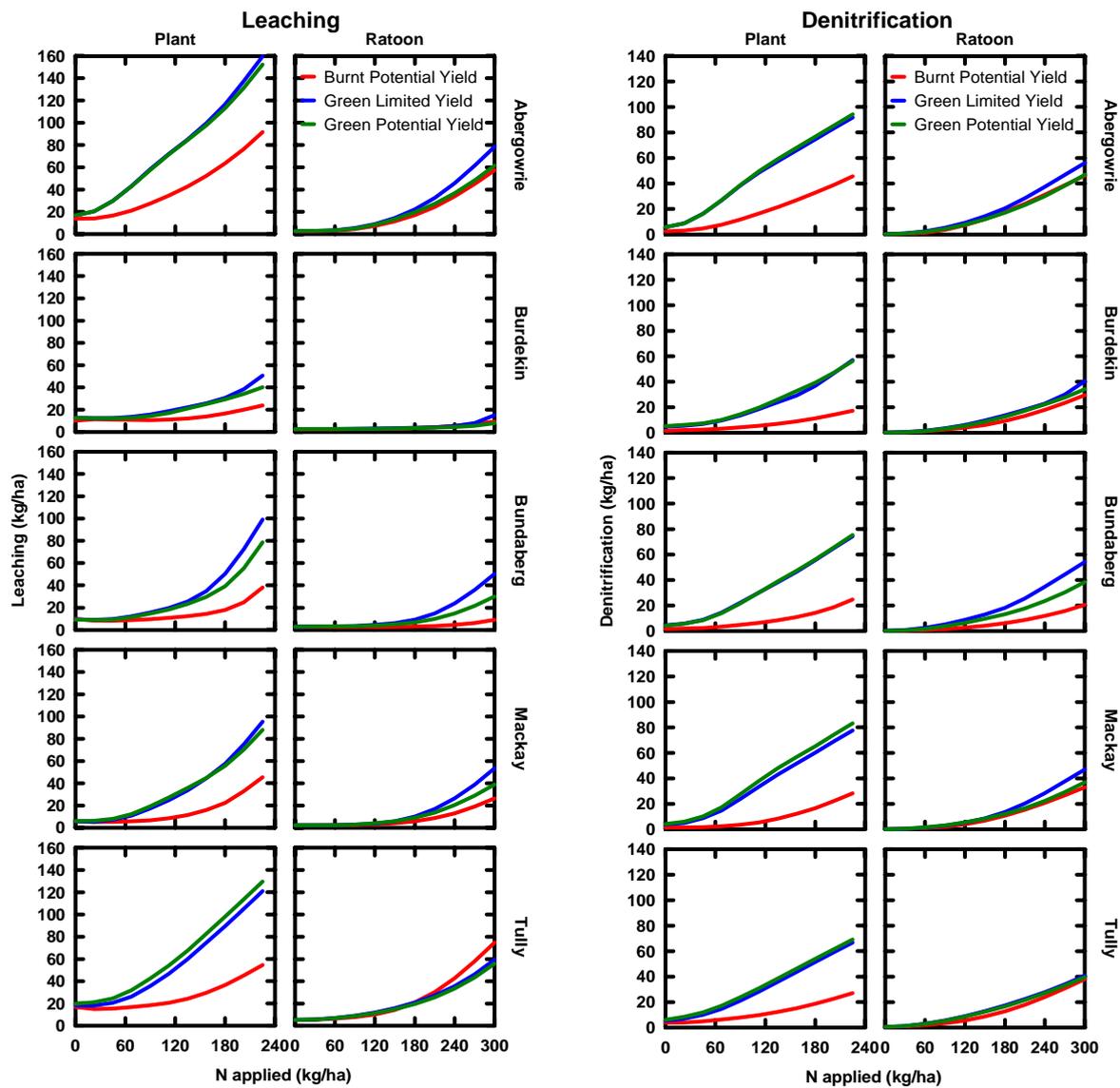


Figure 51. Simulated long-term average rates of nitrate leaching and denitrification in response to applied N fertiliser for different trash management systems for a Red Kandosol soil.

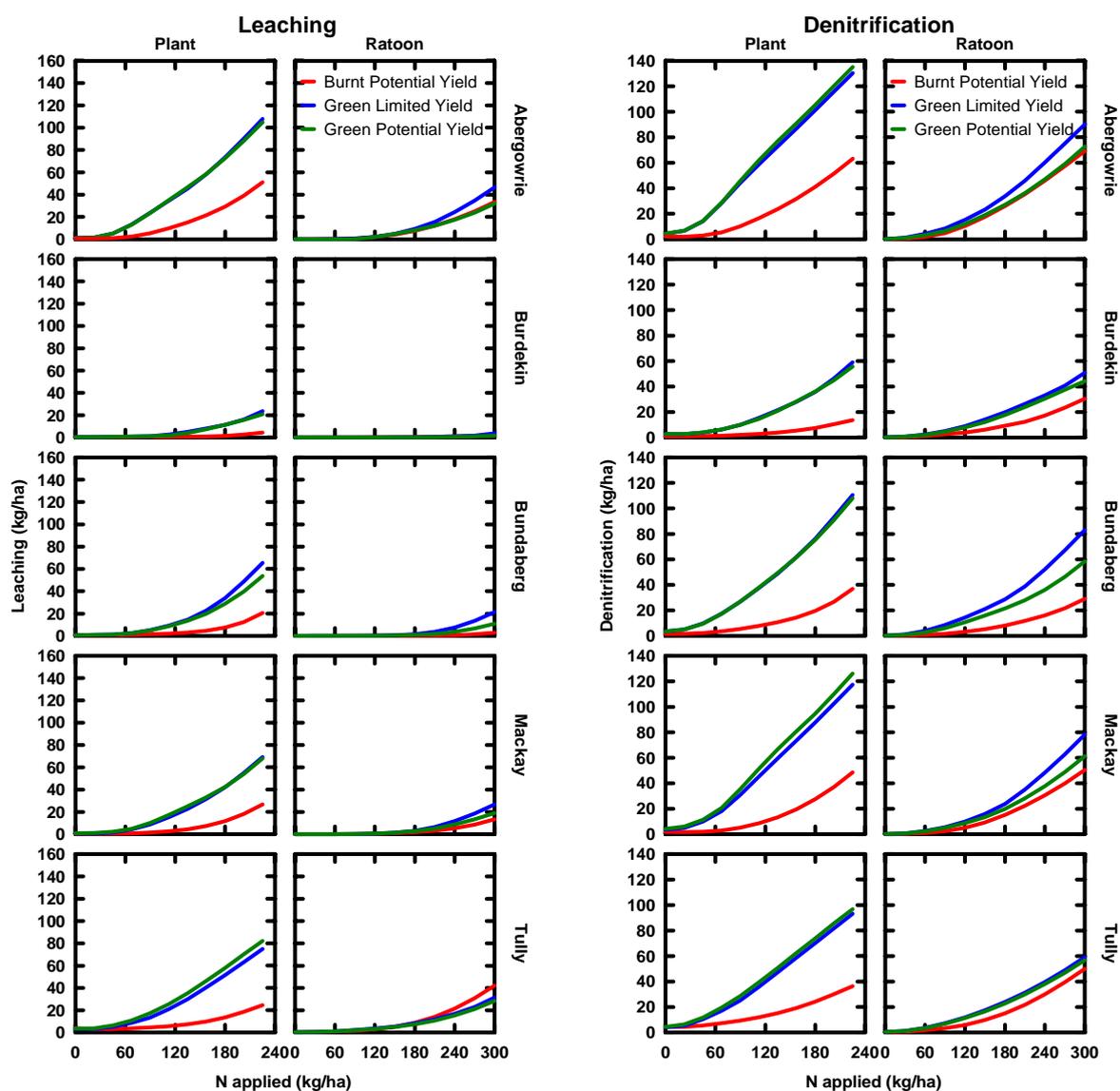


Figure 52. Simulated long-term average rates of nitrate leaching and denitrification in response to applied N fertiliser for different trash management systems for a Brown Chromosol soil.

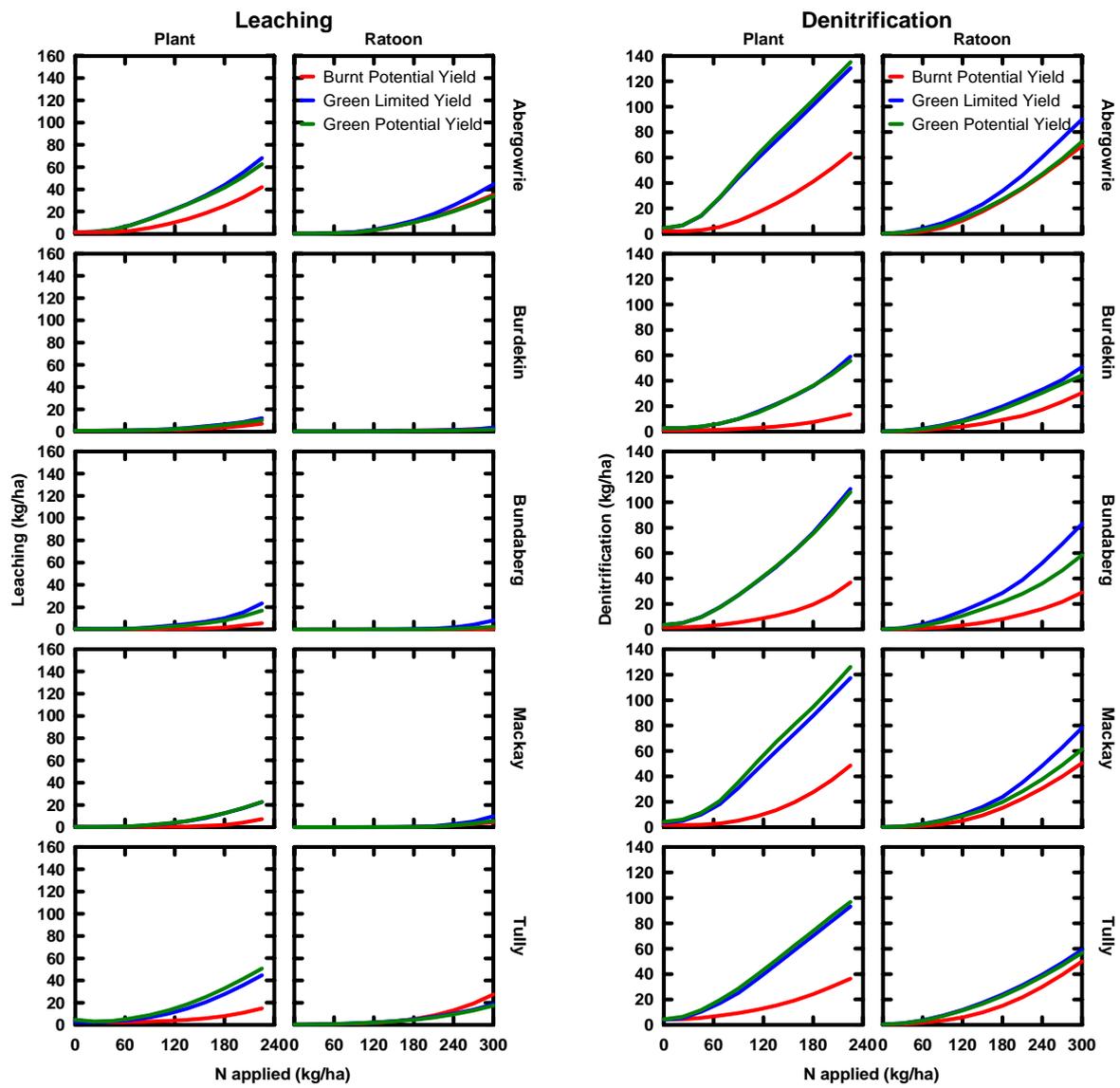
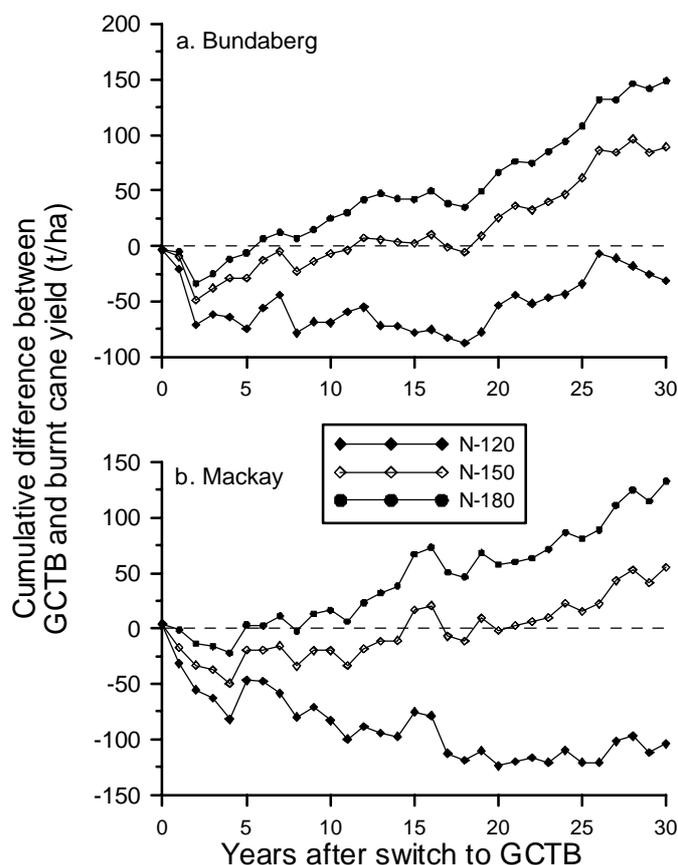


Figure 53. Simulated long-term average rates of nitrate leaching and denitrification in response to applied N fertiliser for different trash management systems for a Brown Dermosol soil.

Short-term impacts of trash

Since decomposing trash blankets have the potential to immobilise considerable amounts of N (Ng Kee Kwong *et al.*, 1987; Basanta *et al.*, 2003), there may be a possibility of N stress developing and limiting sugarcane yields following the switch from burning to retaining trash. To investigate this through the simulations, yields with N applied at optimum rates in the burnt system were compared to those in the trash retained system at various N rates. Examples of this analysis are shown in Figure 54 for the Bundaberg climate with Red Kandosol soil and Mackay climate with Brown Chromosol soil. In these examples, the cumulative difference (increase or decrease) between simulated yields in the trash retained (at various rates of N) and burnt (at optimum N) systems are shown through time. The **cumulative** differences are shown as they remove the impacts of large year-to-year variability (e.g., Figure 47) on depiction of the results. Potential yields are analysed for the trash retained system as this will show the greatest effect of N immobilisation. In both examples the optimum N application rates in the burnt system was taken as 150 kg/ha on ratoon crops (Figure 54).

Figure 54. Simulated change in cumulative difference in sugarcane yield between GCTB (at three different rates of N fertiliser) and trash burnt (at 150 kg/ha application of N fertiliser) systems with increasing time after changing from the burnt to the GCTB system for (a) Bundaberg climate and Red Kandosol soil and (b) Mackay climate and Brown Chromosol soil.



At the rates of N applied to the trash retained systems shown in Figure 54, simulated yields tend to be lower (i.e. the cumulative difference is increasingly negative) immediately following the switch from trash burning to trash retention. The duration of this period of yield depression varies depending on the rate of N; e.g. it is 15 to 20 years with 120 kg/ha of N, to 2 to 4 years at 180 kg/ha of N. It is not until after this time that the yield benefits simulated for trash retention occur, and it may take some time (5-15 years for the examples in Figure

54) for the cumulative yield benefit of the trash retained system to be positive. In the Bundaberg climate with Red Kandosol soil for example (Figure 54a), the yields with trash retained increase relative to the burnt system after four years with 180 kg/ha of N applied to the trash retained system. However, it is not until ~ 20 years that they increase with 120 kg/ha of N.

Discussion

This simulation study has shown that sugarcane production has the potential to respond positively to trash retention in a wide range of environments, from Bundaberg to Tully (Figures 48, 49 and 50), and provided soils have good internal drainage. This benefit will be most pronounced in ratoon crops as sugarcane is rarely planted into a trash blanket. The production benefits of trash blanketing are widely recognised from both experiments (Wood, 1991; Chapman *et al.*, 2001) and anecdotal accounts. However, this study also shows that achieving these production benefits (i.e. higher potential yields) makes use of much of the trash N that is recycled in the soil-crop system. The remainder is lost to the environment (Figures 51, 52 and 53). Therefore, N application rates should not be reduced following the switch from a burnt to GCTB system, provided there is an expectation that cane yields will be higher. Linking N management with yield expectations has not been recognised in previous recommendations of N management following the change to a GCTB system (Calcino *et al.*, 2000; Basanta *et al.*, 2003).

These previous recommendations have implicitly assumed a similar target yield in both burnt and GCTB systems and so, logically, suggested N applications be reduced. There will be situations when yields will not be higher under GCTB, such as in areas with poor drainage (Wood, 1991) or early harvested crop in NSW (Kingston *et al.*, 1998). However, even when yields in a trash blanketed system are similar to those in a burnt system, the simulations undertaken in this study suggest that rates of N fertiliser needed for ratoon crops in GCTB are similar to those in burnt systems (Figures 48, 49 and 50).

It is only in plant crops where there is evidence that N recycled in the trash retained system 'replacing' N from fertiliser (Figures 48, 49 and 50). This study suggests that N rates for plant crops in GCTB systems can be approximately halved in comparison with those in burnt systems. This recommendation is independent of whether or not yields are expected to be increased by trash retention.

This study has also illustrated the potential negative, short-term impact of trash blanketing on sugarcane yields (Figure 54) due to the immobilisation of N by the decomposing trash. While the immobilisation of trash N has been demonstrated experimentally before (Ng Kee Kwong *et al.*, 1987; Basanta *et al.*, 2003) its potential impact on yield has not. As both soil organic matter concentrations and soil N mineralisation rates increase following the change to trash blanketing, there is a sufficient N supply in the soil-crop system to allow the trash blanket to decompose without creating N deficits; i.e. the system comes into a new equilibrium. The results of these simulations suggest that it takes at least 5 years for this equilibrium to be reached, possibly longer if there is less available N. There are interesting implications of these insights into the time to equilibrium (i.e. the period of disequilibrium). Short-term trash management trials have the potential to produce different results depending on the trash management history of the site prior to the trial, whether trash was previously burnt or retained, and the rate of N applied during the trial. If trash at the site was previously burnt, short-term results may reflect this period of disequilibrium rather than the true potential of GCTB at the site. Higher than optimum applications of N to trash trials would also impact the results, minimising the disequilibrium effects in the GCTB treatment. Details of pre-experiment management are seldom given and might explain some of the disparity in results

obtained from many trash trials. It is also interesting to note that the widespread adoption of GCTB in the Australian sugar industry through the 1980's and early 1990's occurred at a time where **average** N fertiliser use in the industry (~ 2 kg N/t cane) was greater than that required for maximum sugarcane production (< 1.5 kg N/t cane). Thus, it is likely that the phase of disequilibrium following the widespread adoption of trash blanketing was minimised by the plentiful N supply. With the recent trend of lower N applications relative to sugarcane yields, there may be more short-term negative experiences for those growers who adopt trash blanketing now.

The results of the simulations also indicate that average environmental losses of N are likely to be greater from GCTB systems at all rates on N fertiliser applications (Figures 51, 52 and 53). The higher simulated rates of denitrification in the GCTB system (Figures 51, 52 and 53) are consistent with those of the short-term study of denitrification conducted in this project (Figures 14 and 15). However, there are no other experimental comparisons of N losses under different trash management systems, especially at different N application rates. Despite the lack of experimental experience to support the simulation results in this area, this study suggests that even more caution should be exercised to avoid over-application of N in GCTB systems.

As noted above, the simulation-based approach taken in this study was necessary because of the dearth of experimental information, both in Australia and overseas, about the interactions between trash management and N fertiliser management in sugarcane production systems. There are many limitations to a simulation-based approach, including the one-dimensional assumptions of the models and process not considered in the model. Others include the climates and soil types not considered in this study. One notable absence in the soil types is soil high in organic C, as the soils represented in this study came from a relatively narrow range (0.8-1.2 %, Table 35). However, a similar, but more limited, analysis undertaken with the parameters for the soil high in organic C (4.2 %) at the BT1 site by Thorburn *et al.* (2002) produced very similar patterns in results, suggesting the results of this study may have some generality. Another possible limitation is that the change from potential yields in the simulations to actual yield obtained in the field may change the implications for N management in a GCTB system. An example of this may be lodging, water logging and other factors restricting yields, so that there is little difference in those in burnt and GCTB systems. We have attempted to represent this situation empirically in the simulations by restricting radiation use efficiency in the sugarcane model, but this may not fully capture the impacts of trash and N fertiliser management in the field in the presence of lodging, water logging, etc. This limitation, together with those listed above suggest that it would be still valuable to obtain experimental data on the interactions between trash and N fertiliser management in a wide range of environments.

13. INDUSTRY AWARENESS OF PROJECT RESULTS

There were numerous opportunities for raising industry awareness of the results of this project. Many of these were *ad hoc* opportunities that were taken, such as delivering presentations at field days, shed meetings, Mill Supplier Committee meetings, CRC-Sugar meetings (at which BSES extension officers were present) and ASSCT. However, more strategic activities were also undertaken, and these will be the focus of this report.

Substantial effort was directed to conducting a series of workshops, in collaboration with the CRC-Sugar, on the impact of trash blanketing on soil fertility and fertiliser management. The workshops were organised and presented by the project Principal Investigators at that time (Peter Thorburn and Ross Mitchell) in collaboration with CRC-Sugar researcher Fiona Robertson and CRC-Sugar Extension Officer Ingrid Christensen. The workshops were aimed at BSES, productivity board and fertiliser company advisers, and were held in Townsville (28/6/00), Cairns (29/6/00) and Bundaberg (8/8/00). A summary of results from this project and associated work in CRC-Sugar was presented at the workshops. Participants then developed strategies for incorporating the information into local fertiliser application recommendations.

The workshops were conducted using a different approach from the normal “university lecture” style. The new approach was a *participatory approach to information exchange*, based on adult learning principles. Information on the current state of research was presented to advisory staff and they were invited to contribute to the development of locally-relevant “best-bet” management strategies. In this way, the development of best management strategies draws not only on research knowledge but also on advisers’ experience, lending a far greater degree of ownership and understanding of the recommendations among the advisers. Each workshop was also **purposefully** limited to ½ a day, to increase the chances of participation. Informal surveying of potential participants suggested that participation in longer (1 or 2) workshops was too disruptive to work schedules, and people were much less likely to attend (particularly during harvest).

The workshops were evaluated, gaining feedback on the content, presentation and format. The evaluation questionnaire is shown in Appendix 1.

The workshops conducted in collaboration with the CRC-Sugar were very successful. There were 20-30 participants at each of the three workshops. Over 95 % of all participants *agreed* or *strongly agreed* that the workshops were (1) ‘informative’ and (2) ‘worthwhile attending’ (a complete evaluation report is given in Appendix 2). Approximately 70 % *agreed* or *strongly agreed* that the ‘information could be readily applied when giving advice on fertiliser management’ and more than 60 % *agreed* or *strongly agreed* that they would have ‘more confidence when giving that advice’. There was little consistent difference in ratings between the three workshops, although there was a trend towards greater proportions of positive responses to the above questions in the later workshops, when presentation content was refined. Participants also rated the workshop format and duration highly.

The local TV and radio press broadcast interviews with members of the workshop team.

14. OUTPUTS

General

There are several outputs arising from the project. One is the volume of data generated during the project on many properties of trash, its dynamics and impacts. The vast majority of this information was not known before the project. From these data, we now have a greatly improved insights into many aspects of nutrient cycling in trash blanketed systems. Many of these data and associate understandings have been documented in publications (listed below), which are widely available.

Another output was the modelling developments made during this project. These include improvements in the description of crop residue decomposition in APSIM-Residue, that allow trash decomposition to be predicted accurately, and new parameter values for denitrification algorithms in APSIM-SoilN. Experience gained in this project has resulted in some changes to the initialisation of parameters in the APSIM-SoilN model compared to the approaches used in previous N modelling studies in the sugar industry. These outputs are readily available to the sugar industry and can be used in further research and advisory activities.

During the course of this project, much information has been gained on the properties of soils at various research sites in the industry. Future experiments conducted on these sites (i.e. those on BSES Ltd Research Stations) will benefit from this information. In addition, this information provides a useful database from which values for model parameters can be and have been developed.

Publications arising from this project

Journal papers

- Thorburn, P.J., Probert, M.E. and Robertson, F.A. (2001). Modelling decomposition of sugarcane surface residues with APSIM-Residue. *Field Crops Research*, 70: 223-232.
- Thorburn P.J., Meier E.A. and Probert M.E. (2005). Modelling nitrogen dynamics in sugarcane systems: Recent advances and applications. *Field Crops Research*, in press.

Book chapters

- Robertson, F.A. and Thorburn, P.J. (2001). Crop residue effects on soil C and N cycling under sugarcane. In: Rees R.M., Ball B.C., Campbell C.D. and Watson C.A. (Eds), *Sustainable Management of Soil Organic Matter*, CAB International, Wallingford, UK, pp 112-119.
- Thorburn, P.J., Robertson, F.A., Lisson, S.N. and Biggs, J.S. (2001). Modelling breakdown of sugarcane residues and the impact on soil nitrogen, carbon and water. In: Rees R.M., Ball B.C., Campbell C.D. and Watson C.A. (Eds), *Sustainable Management of Soil Organic Matter*, CAB International, Wallingford, UK, pp 74-82.
- Probert M.E., Keating B.A., Thorburn P.J. and Lisson S.N. (2000). Management options for carbon sequestration in tropical and sub-tropical cropping systems. In: Keenan R., Bugg A.L. and Ainslie H. (Eds), *Management options for carbon sequestration in forest*,

agriculture and rangeland ecosystems. Cooperative Research Centre for Greenhouse Accounting, Canberra, pp 38-48.

Refereed conference papers

- Thorburn P.J., Probert M.E., Lisson S., Wood A.W. and Keating B.A. (1999). Impacts of trash retention on soil nitrogen and water: an example from the Australian sugarcane industry. *Proceedings of the South African Sugar Technologists' Association*, 73: 75-79.
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- Thorburn, P.J., Van Antwerpen, R., Meyer, J.H. and Bezuidenhout C.N. (2002). The impact of trash management on soil carbon and nitrogen: I Modelling long-term experimental results in the South African sugar industry. *Proceedings of the South African Sugar Technologists' Association*, 76: 260-268.
- Thorburn, P.J., Biggs, J.S., Horan, H.L. (2003). Modelling the impact of sugarcane crop residue management on soil nitrogen management. In: *Solutions for a Better Environment*. Proceedings of the 11th Australian Agronomy Conference, February 2003. Geelong, Victoria. Published on CDROM ISBN 0-9750313-0-9.
- Thorburn, P.J., Horan, H.L. and Biggs, J.S. (2004). The impact of trash management on sugarcane production and nitrogen management. *Proceedings of the Australian Society of Sugar Cane Technologists* 26: on CD.

Conference abstracts

- Robertson F.A. and Thorburn P.J. (1998). Decomposition of the green cane trash blanket. *Proceedings Australian Society Sugar Cane Technologists* 20: 553.
- Thorburn P.J., Lisson S.N., Probert M.E., Keating B.A. and Wood A.W. (1999). Plant residue retention in sugarcane systems: Modelling impacts on sustainable production. In: *Food and Forestry: Global changes and global challenges. The GCTE Focus 3 Conference delegates manual* (Elsevier Science, Oxford), p P70.
- Thorburn, P.J., Robertson, F.A. and Keating, B.A. (2001). Trash management – consequences for nitrogen management and cane productivity. *Proceeding Australian Society Sugar Cane Technologists*, 23: 474.
- Thorburn, P.J., Biggs, J.S. and Horan, H.L. (2003). The impact of trash management on soil nitrogen management. *Proceedings of the Australian Society of Sugar Cane Technologists* 25: 1p.

Awards for work conducted in this project

Best Agricultural Poster and Poster Presentation – Australian Society of Sugar Cane Technologists Conference, May 2001 (Thorburn, P.J., Robertson, F.A. and Keating, B.A., Trash management – consequences for nitrogen management and cane productivity).

Best Presentation – CRC-Sugar Annual Conference, March 2001 (Thorburn, P.J., Keating, B.A., Robertson, F.A., Probert, M.E., Biggs, J.S. and Lisson, S.N., Impact of trash management on nitrogen cycling).

Kynoch Prize (Best Agricultural Paper) – South African Sugar Technologists' Association Conference, August 2002 (Thorburn, P.J., Van Antwerpen, R., Meyer, J.H. and Bezuidenhout C.N., The impact of trash management on soil carbon and nitrogen: I Modelling long-term experimental results in the South African sugar industry).

15. EXPECTED OUTCOMES

This project has had, and will continue to have several outcomes. However, the main outcome has been the insights gained into nutrient cycling, especially for N and K, in sugarcane production systems. There is already evidence of the impact of knowledge gained in this project. Modelling developments made in this project are being used in other projects in the Australian sugar industry (e.g., PhD projects, CSE008, CSE009, CSE010, CSE011, CSE012) to provide insights into topics such as new approaches to management of N in the sugar industry, management practices to improve the quality of water leaving sugarcane farms and the value chain impact of using trash to fuel co-generation plants. Future outcomes will centre on the use of the information gained in this project in the development of improved nutrient management schemes in the sugar industry. This will ultimately benefit the industry through better matching of nutritional inputs to requirements, maximising farm profitability, maintaining the resource base and minimising off-site impacts.

16. FUTURE RESEARCH NEEDS

This project was conceived and, generally, conducted at a time when industry-wide nutrient recommendations were the norm. Recent advances in nutrient management (Wood *et al.*, 2003; Thorburn *et al.*, 2003a) have been towards soil- or block-specific and yield-specific strategies. Thus the general recommendations made in this project will (and should) be superseded by the more situation-specific management practices. However, the recent advances do not explicitly consider the effect of different trash management practices on nutrient management recommendations. Given the importance of trash in K supply and N cycling, nutrient management strategies developed for the sugar industry should explicitly consider the different ways of managing trash.

While results from field and pot experiments in this project confirm the plant availability of potassium released from sugarcane residues, we are not recommending an equivalent reduction in potassium fertiliser inputs. This caution is suggested because of the evidence that while current recommendations for potassium nutrition are optimised for yield performance, there is also increasing evidence that such systems are not exploiters of soil potassium. Therefore more systems based research is required to guide the adoption of sustainable techniques for management of potassium fertility in soils.

Many of the insights into N fertiliser management obtained in this project have arisen from simulation-based analyses. This approach was necessary because of the dearth of experimental information on the interactions between trash and N fertiliser management. It would be still valuable for the sugar industry to obtain experimental data on these interactions to provide an independent test of the implications of the simulation results. This included more detailed studies of biologically driven process, e.g. trash decomposition and denitrification, as affected by management practices that significantly impact on soil biology (such as biocide use, time under trash blanketing, fallow management, etc.).

These simulations have suggested that even more care needs to be taken to minimise environmental losses of N from GCTB sugarcane production systems. Yet, no measurements have been made of environmental losses of N at a range of N fertiliser rates, in either burnt or GCTB systems. These data are necessary to confirm the implications of simulations undertaken in this study and others of the relationship between environmental losses of N and N fertiliser rates.

17. RECOMMENDATIONS

The main recommendations from this study centre on the changes to fertiliser management strategies that should be adopted for GCTB systems. These are given below for the different types of nutrients. However, the recommendations should be interpreted in light of the need for additional research described in the previous Section.

Nitrogen

Simulation studies of N dynamics in GCTB systems suggest that:

- For at least 5 years after adoption of GCTB, it is important that N applications are not reduced below those used when trash was burnt.
- After this time, N fertiliser applications to *plant* crops in GCTB systems could be reduced to half that recommended for burnt systems.
- N fertiliser applications to *ratoon* crops in GCTB systems should be maintained at rates appropriate for burnt systems.
- Particular caution should be taken to avoid over-application of N in GCTB systems to minimise environmental losses of N, from both denitrification and leaching.

Potassium

While K from trash was shown to be plant-available, it is recommended that K management in GCTB systems should be similar to those in burnt systems.

Other nutrients

The availability of other nutrients in GCTB systems was not greater than in burnt systems, so nutrient management should not be changed after adoption of GCTB.

18. REFERENCES

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APPENDIX 1.**Evaluation forms distributed at trash research information workshops****WORKSHOP EVALUATION****GCTB and Nutrient Management****LOCATION, DATE****Your affiliation:** BSES CPPB Agribusiness Other

	Strongly Disagree			Strongly Agree	
Overall the workshop was informative	1	2	3	4	5
I can readily apply the information when giving advice	1	2	3	4	5
It has given me greater confidence to make fertiliser recommendations for GCTB	1	2	3	4	5
There was nothing new presented/discussed	1	2	3	4	5
<i>The workshop improved my understanding of:</i>					
Research underway on nutritional aspects of GCTB	1	2	3	4	5
Significance of this research	1	2	3	4	5
What/how much nutrients are contained in trash	1	2	3	4	5
Availability of nutrients from the trash blanket	1	2	3	4	5
How to adjust fertiliser recommendations for GCTB	1	2	3	4	5
Complexity of nutrient cycling	1	2	3	4	5
Factors influencing nutrient cycling	1	2	3	4	5
Where nutrients go when cane is burnt	1	2	3	4	5
It was worth my time to attend	1	2	3	4	5
The speakers went into sufficient detail	1	2	3	4	5
The speakers were clear to understand - Ross	1	2	3	4	5
Fiona	1	2	3	4	5
Peter	1	2	3	4	5
The group session helped clarify the information.	1	2	3	4	5
The group sessions helped me to apply the information to my situation.	1	2	3	4	5
Discussing experiences and advice with other attendees was valuable.	1	2	3	4	5
I would have gained just as much from reading a report/paper on the information	1	2	3	4	5

Key messages I have gained from the workshop are:

.....
.....

I would have liked to hear more about:

.....
.....

The workshop could be improved by:

.....
.....

Please rank preferences:

I prefer to hear this type of information:

- In this short workshop format.
- As a part of a 2-3 day course.
- At ASSCT.
- As a written report/paper.
- Media reports (Bulletin, etc)
- Via a website
- Other

I would like to hear updated information via:

- Email
- Newsletter
- BSES Bulletin
- Australian Sugarcane
- ASSCT
- CRC Website
- BSES Website
- Other

Any other comments:

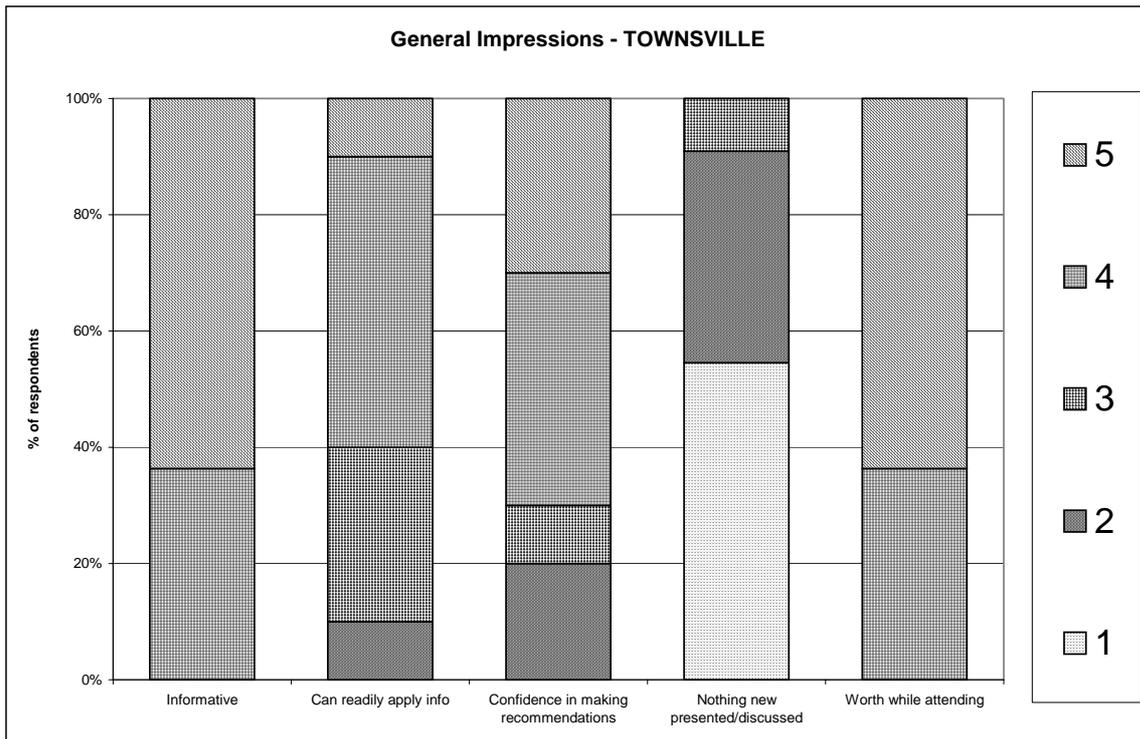
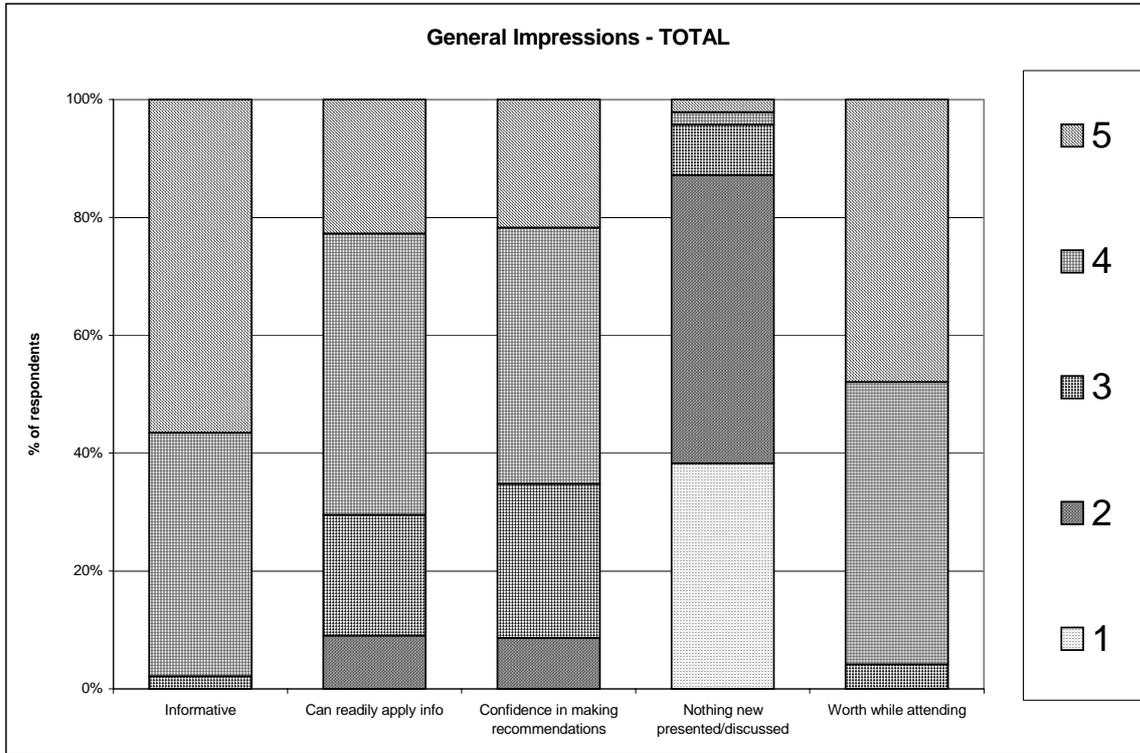
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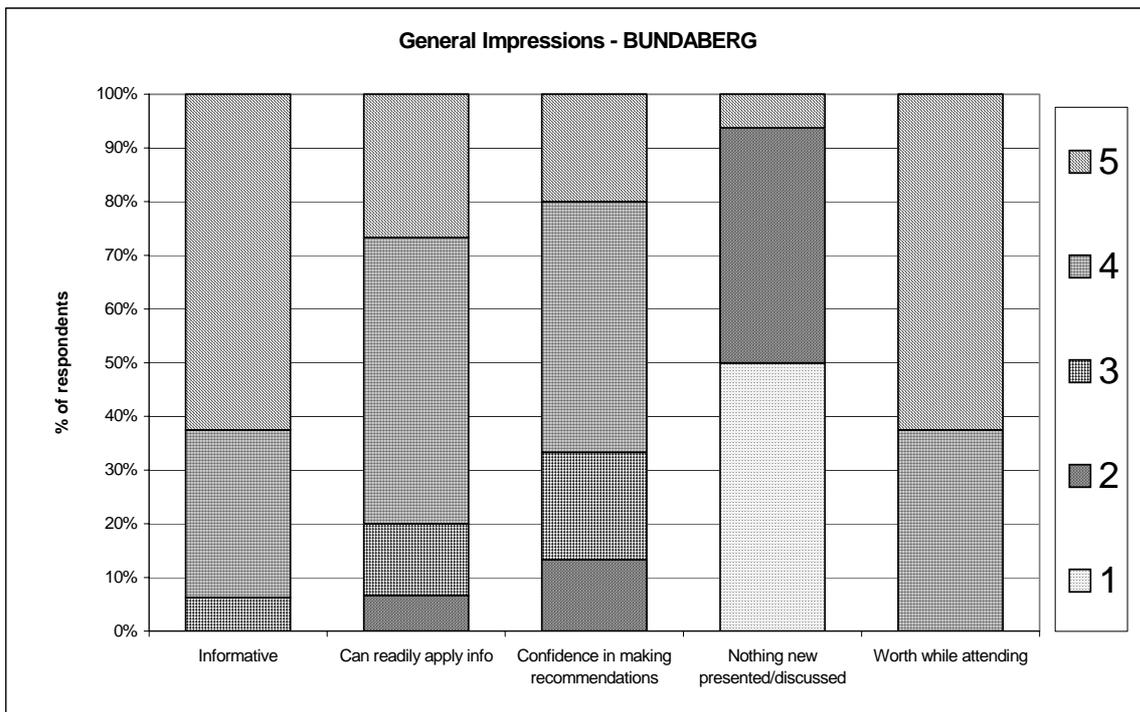
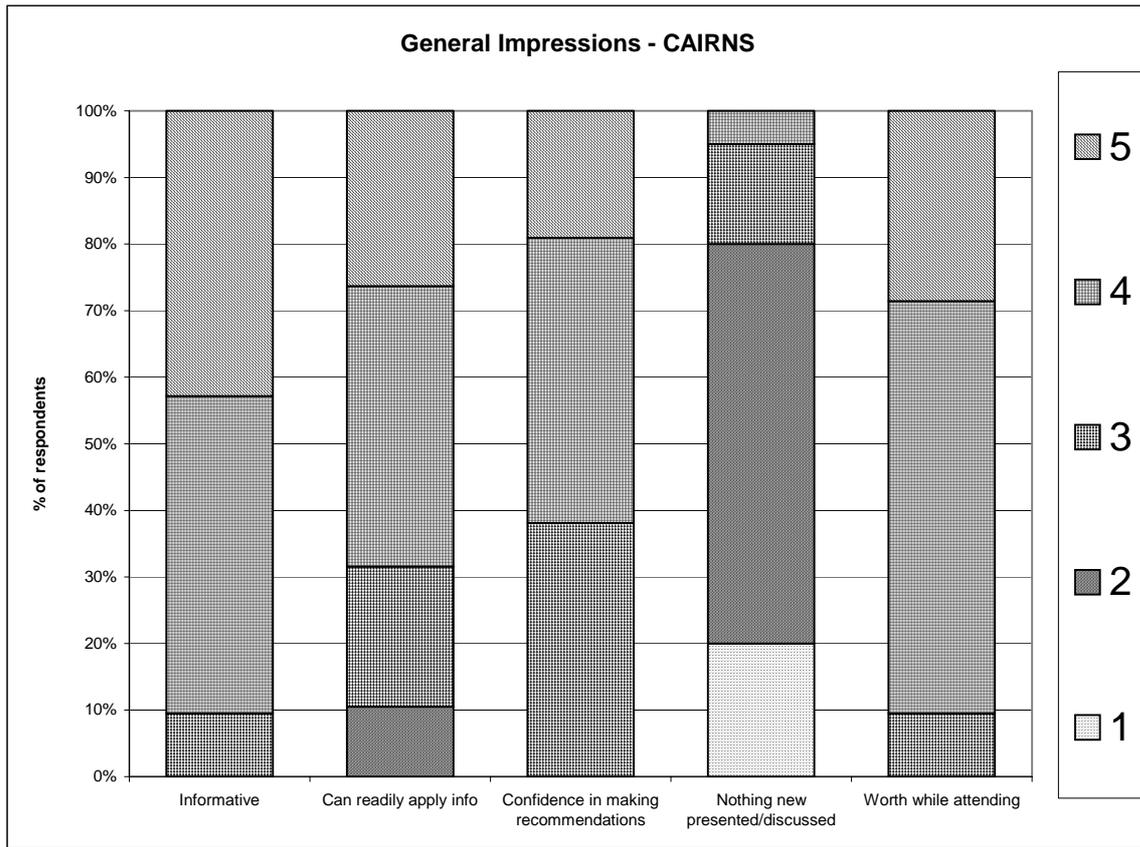
Thank-you!

APPENDIX 2.

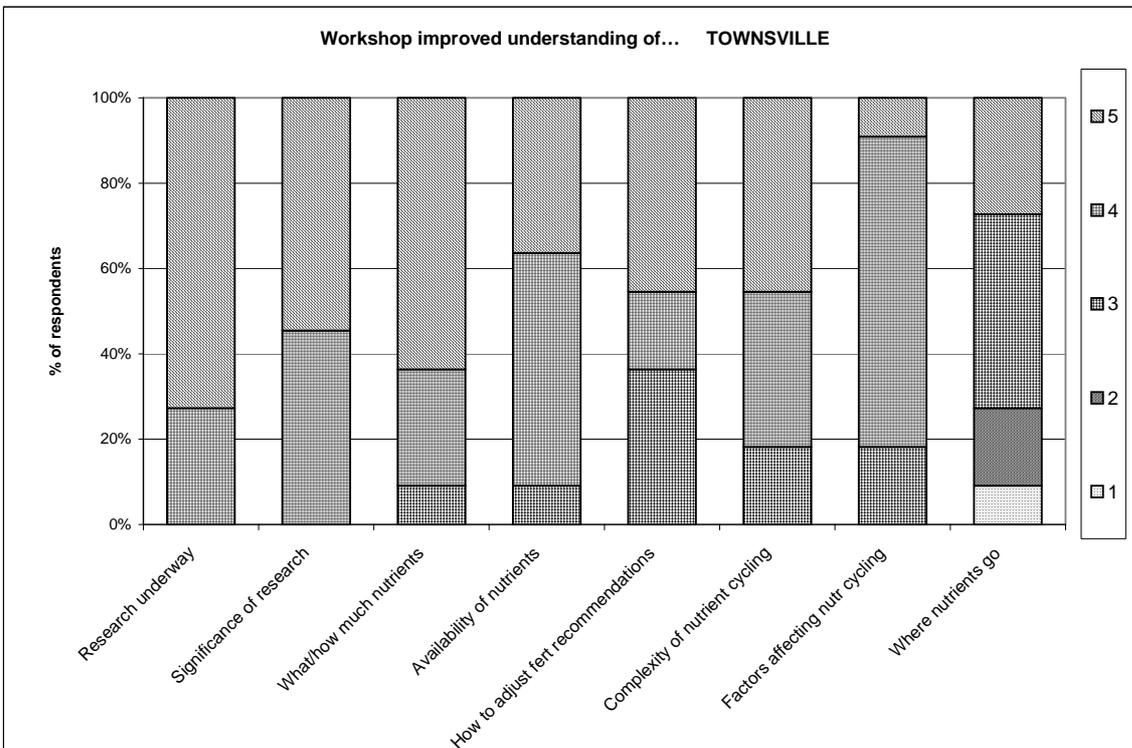
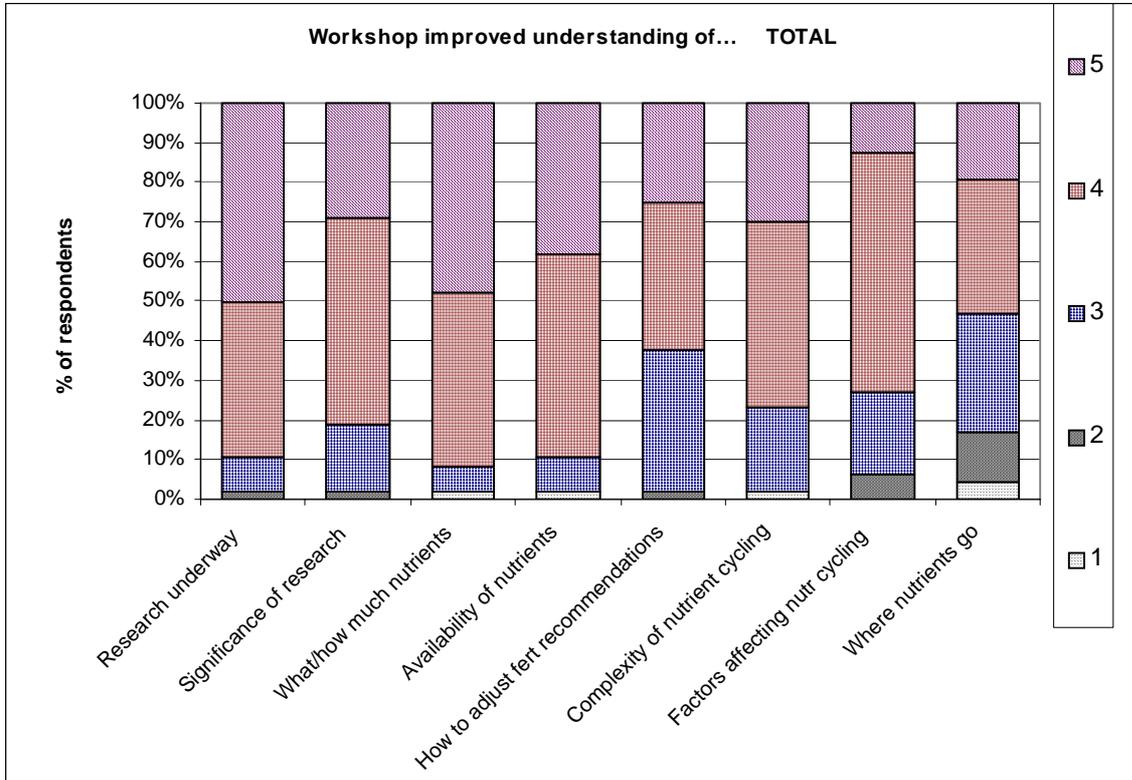
Results of evaluation of trash research information workshops

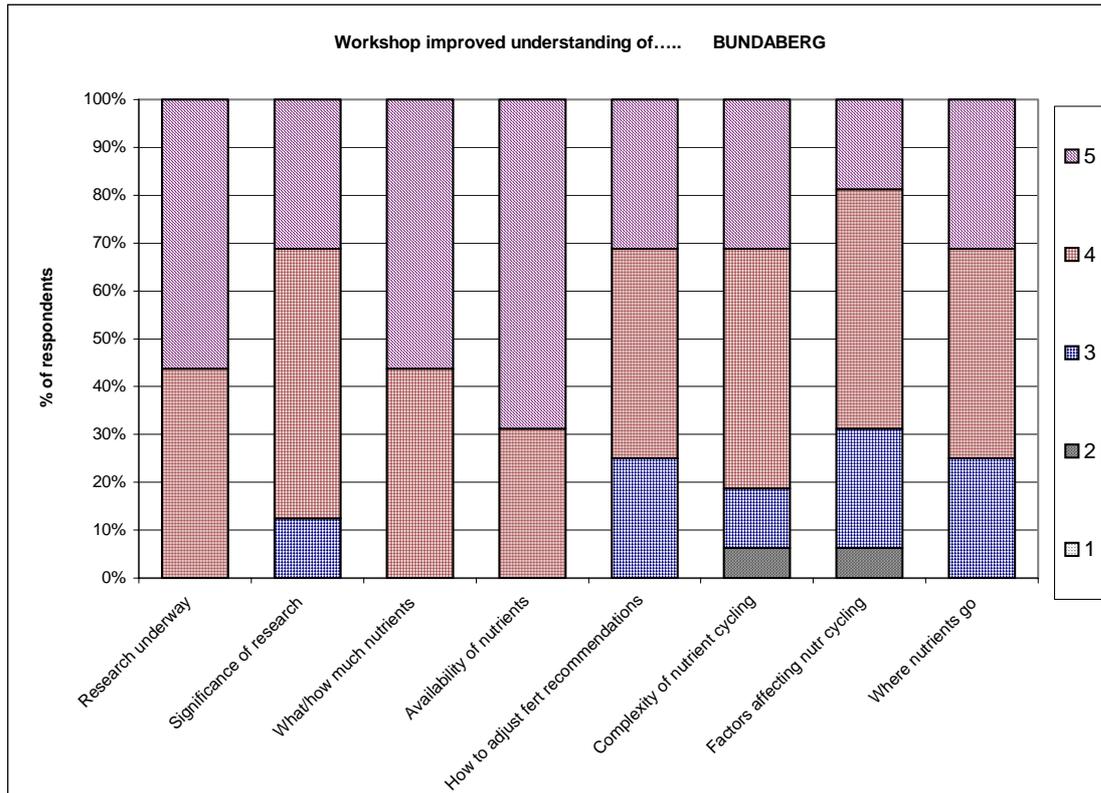
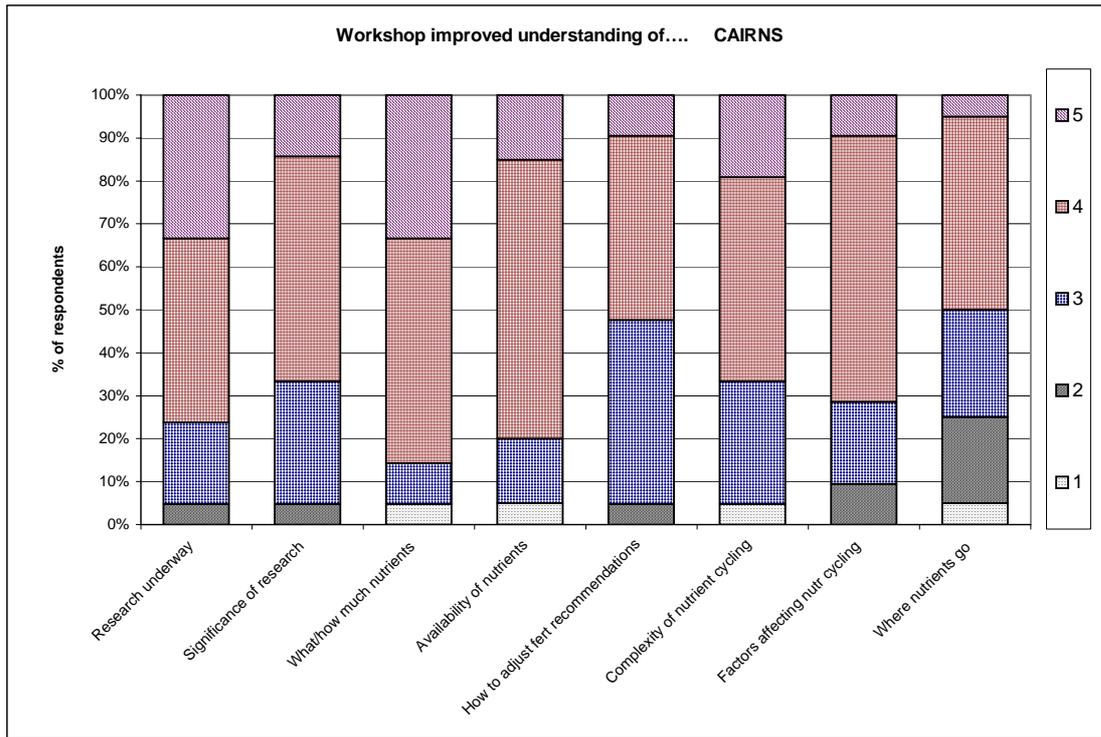
General impressions about the workshop?





Has the workshop improved the participants understanding of issues related to GCTB and this research?





Quality of presentations and value of group activities?

