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Impacts of fertilisers and legumes on N_2O and CO_2 emissions from soils in subtropical agricultural systems: A simulation study

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ABSTRACT

There is increasing focus on greenhouse gas emissions from agricultural systems. One suggested method for increasing the sequestration of carbon (C) within agricultural soils is to increase crop productivity and therefore C input into the soil. However, if enhanced production is achieved via nitrogenous fertilisers, there is a potential tradeoff between decreased C emissions and increased nitrous oxide (N_2O) emissions due to the increased soil C and nitrogen (N). An alternative is to incorporate leguminous crops into cereal cropping rotations to provide a biological source of N. However, the likely production of N_2O from N released during the decomposition of leguminous residues is unknown as is the impact on C input into the soil when some cereal crops are replaced with grain legumes. Consequently, an analysis of the likely impacts has been undertaken for a subtropical dryland cropping system in Queensland, Australia where soil, climate and management are conducive to denitrification losses.

A series of scenarios embracing a range of cropping rotations, N fertilisers and leguminous crops was tested using the Agricultural Production Systems Simulator (APSIM). The model configuration was tested using long term data from the Brigalow Catchment Study site near Theodore, Queensland, Australia (24.81°S, 149.80°E). A wide range of data was used in testing the model for the major terms in the C, N and water balances.

Scenario analyses of alternative management systems including the use of fertiliser or legume grain or forage crops within cereal rotations demonstrated that soil C can be managed to some degree via simple changes in agronomic practice. The use of legumes within cereal rotations was not always as effective in reducing N_2O emissions as improved fertiliser practice. For example, replacing wheat with chickpea did not reduce N_2O emission relative to fertilised systems and did not assist in increasing soil C due to impacts on stubble cover over the important summer months. The fact that some interventions proved counterproductive due to complex feedback mechanisms highlights the need for detailed models which capture the links between water, C, N and management.

1. Introduction

There is growing focus in many agricultural systems around the globe on increasing soil organic matter content for various reasons including increased fertility, improved soil health and, more recently, carbon (C) sequestration for greenhouse gas mitigation. Enhancing soil C, or at least maintaining existing levels, requires an increased input of C into the soil via retention of crop residues, improved crop growth, or increased cropping frequency (Wang et al., 2004; Franzluebbers, 2005). If it is achieved via increasing crop yield or frequency, an increase in nutrient input will be required. However, this increases the risk of adverse outcomes such as soil acidification or nitrate leaching into streams or ground waters.

There is also the risk that increased production of nitrous oxide (N_2O) during denitrification might offset some of the greenhouse gas benefits of C sequestration. This paper illustrates the mechanisms by which this can occur in subtropical agricultural systems.

This increased risk of greenhouse gas emissions is difficult to quantify empirically because it is influenced by a wide range of environmental and management factors. Climate, soils and management act together in complex ways to determine the impact of C management on nitrogen (N) losses and the interactions are not well understood. For example, increased N

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fertiliser inputs resulted in increased N_2O emissions for silty and sandy loam soils in Uzbekistan (Scheer et al., 2008) but made little impact on sandy soils in Western Australia (Barton et al., 2008). The influence of various N sources is also inconsistent. For example, losses of N derived from legumes were lower than for those from fertiliser in cotton-based systems (Rochester et al., 2001) but not when growing legumes with cereal crops (Dick et al., 2008). Detailed studies for the south-eastern United States (Franzluebbers, 2005), eastern Canada (Gregorich et al., 2005) and Australia (Rochester, 2003) have all found the level of information available on changes in N_2O emissions in response to management to be inadequate. The same is true for many of Australia's subtropical farming systems.

In Australia's subtropical dryland cropping systems levels of organic matter have decreased significantly since extensive clearing of native woodlands and forests for agriculture. More than 60% of surface soil organic matter can be lost after 50 years of cereal cropping (Dalal and Chan, 2001) and extensive research has been undertaken into reducing this rundown through stubble retention, fertilisers, legumes and pasture phases. However, in these regions, the combination of a subtropical climate and large areas of heavy clay soils predispose such interventions to increasing the risk of N_2O emissions (Weier et al., 1993). The area is also strongly influenced by ENSO (El Niño Southern Oscillation) cycles with extreme wet and dry periods which lead to water logging conditions, or periods of drought that result in extended fallow conditions, both of which affect C and N losses. Moreover, the cropping systems developed by land managers to minimise the impacts of this climatic risk do so by maximising the storage of water and N in good quality clay soils (Thomas et al., 2007) and seem likely to further exacerbate the N_2O losses. However, the magnitudes of these impacts are unknown.

To clarify this, we undertook a study to quantify the impacts of various C management strategies on N_2O emissions for a site within the northern grain growing region of Australia. A series of scenarios that embraced a range of cropping rotations, N fertilisers and leguminous crops was tested using a detailed farming systems model. A simulation approach was used because of the scarcity of data on greenhouse gas emissions from soil under cereal cropping (Barton et al., 2008), especially for fertile clay soils in the semi-arid tropics. The model was configured and tested using a long-term (25 years) dataset. We use the resultant model configuration to shed light on the likely impacts of C management on greenhouse gas emissions.

2. Model testing

The model used in this analysis was the Agricultural Production Systems Simulator (APSIM) (Keating et al., 2003). APSIM's component-based design allows individual models to interact via a common communications protocol on a daily time step. Models are available for many major crop, pasture and tree species as well as the main soil processes affecting agricultural systems (e.g. water, C, N and phosphorus dynamics, and erosion) including denitrification and N_2O emissions (Probert et al., 1998; Thorburn et al., 2010). APSIM also provides a flexible agricultural management capability enabling the user to specify complex crop rotations and land management regimes. As a result, APSIM has been used in a wide range of studies of fertility management involving fertiliser and tillage (Probert et al., 1995), legumes (Turpin et al., 1997) and greenhouse gas emissions (Thorburn et al., 2010). APSIM Version 7 was used for all simulations in this analysis.

Testing of the APSIM modelling capability was undertaken using the detailed data from the cropped catchment within the Brigalow Catchment Study (BCS) (Cowie et al., 2007) near Theodore, Queensland, Australia (24.81°S, 149.80°E). This study had been established to investigate, amongst other things, the change in catchment water balance and decline in soil fertility after clearing of native Brigalow (*Acacia harpophylla*) forest. Brigalow is a leguminous tree, and soils within these forests contain large amounts of C and N. The dataset includes crop production and organic matter decline (Radford et al., 2007), runoff (Thornton et al., 2007) and deep drainage and chloride leaching (Silburn et al., 2009). Whilst no information on N_2O emissions is available for the study, the existing data provide a good test of the model's ability to predict both the main determinants of denitrification losses and the other major terms of the C and N balance. The BCS includes data for three soil types occurring within three catchments with contrasting land use (uncleared, pasture, cropping). To simplify this analysis, only the most common soil type (upper clay)(see Cowie et al., 2007) within the cropping catchment has been used. The cropping catchment was cleared in 1982 with the first crop being planted during the summer of 1984. Crops during the period to March 2005 were wheat (*Triticum aestivum*) and sorghum (*Sorghum bicolor*) and no fertiliser was applied.

2.1. Method

Configuration of the model was undertaken using a wide range of available information. Agronomic records of sowing dates, cultivar selection, plant populations, tillage and weed spraying were used to reproduce the historical management. Long term soil moisture measurements were used to infer soil hydrological parameters (Fig. 1). Long term air temperature and solar radiation data for the Brigalow Research Station (Australian Meteorological Bureau Station Number 035149) was combined with rainfall from the catchment monitoring station. Data for the average runoff water N concentration from across a 5-year time period was used to calculate N losses in runoff water.

The APSIM-SoilN model includes pools to account for fresh organic matter, microbial biomass, humic and inert C within the soil (Probert et al., 1998). All default parameters (Probert et al., 1998) describing the rates and efficiencies of C flows between pools were retained as these had been previously tested on relevant datasets for the long term impacts of fertilisers, tillage and fallow management, varying organic matter amendments, or legume crop phases (Turpin et al., 1997; Probert et al., 1998; Keating et al., 2003). These standard parameters were also used to simulate observed soil N dynamics on 3 farms within the study

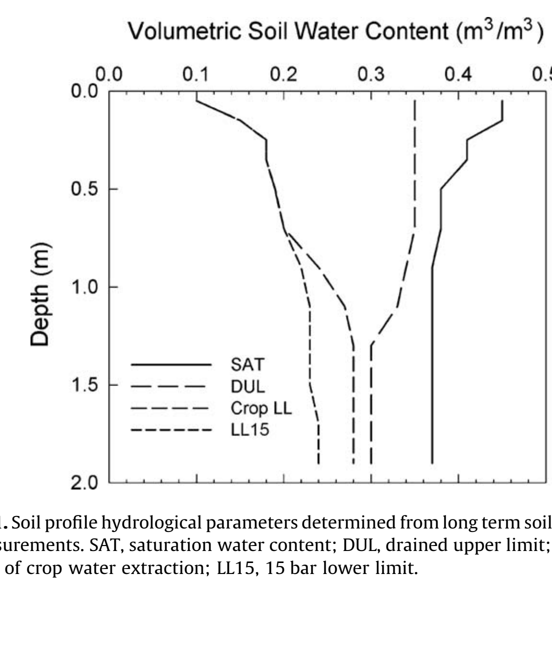

Fig. 1. Soil profile hydrological parameters determined from long term soil moisture measurements. SAT, saturation water content; DUL, drained upper limit; LL, lower limit of crop water extraction; LL15, 15 bar lower limit.

region (Foale et al., 2004). The only alteration to APSIM-SoilN was to increase the value of the denitrification rate coefficient following following Thorburn et al. (2010), who recommended this change after applying numerical optimisation techniques to predictions of measured daily denitrification losses on a similar soil type. The algorithms of Del Grosso et al. (2000), as implemented by Thorburn et al. (2010), have been used to calculate the N_2O component of the predicted total denitrification loss.

Parameterisation of these soil organic matter pools followed a multi-step process making use of data from a variety of sources. Long term soil C data are available for the BCS (Radford et al., 2007) to a depth of 0.3 m (0–0.1, 0.1–0.2, 0.2–0.3 m). This is the depth to which most roots are found in Brigalow forests (Russel et al., 1967) and to where the majority of C is lost after clearing (Harms and Dalal, 2003). Soil C measurements obtained using the method of Walkley and Black (1934) were corrected for incomplete recovery of C using a correction factor of 1.3 (Skjemstad et al., 2000). Resultant values of total C for 0–0.3 m were in good agreement with those obtained using a Leco combustion furnace (C-144) (Skjemstad et al., 2004). Total soil C was partitioned into pools so as to reproduce the two main emergent behaviours of the soil C and N during the BS: (1) a rapid (~9 years) early phase of C decomposition after clearing of the forest followed by a longer slower decline, and (2) a steady increase in soil C:N ratio over time. The soil C lost during the rapid phase of decomposition would, by definition, need to be assigned to the faster soil pools. It is assumed that this C would be mostly in the form of fresh organic matter in natural systems. The increase in soil C:N ratio is likely to be the result of changes in soil C composition from a system dominated by low C:N ratio labile soil pools, to a state consisting mostly of more resistant or inert pools with higher C:N ratios (Skjemstad et al., 2001). The process therefore was as follows:

(i) We assumed that the majority of soil C during the period of slow C decline resides within the inert and humic pools. Crop residues decompose rapidly in these systems (Wang et al., 2004) and microbial biomass constitutes a small fraction of the total C. If we further assume, that the inert C can be represented by measurements of charcoal C (Skjemstad et al., 2004) and that the N content of this pool is low (Hopmans et al., 2005), we can estimate the size and C:N ratio of the humic pool during this later period of decline as the bulk of the N would be contained in the humic pool. Charcoal C was partitioned between the surface layers such that the resultant C:N of the humic pool in each layer was similar. This resulted in an average C:N for the humic pool of 12.8 and this was subsequently applied across the entire soil profile.

(ii) The C:N of the humic pool remains constant within the APSIM model as does the amount of inert C. This being the case, the partitioning of the initial soil C can be performed on the basis of the relative value of the bulk soil C:N and the C:N for the humic and fresh organic matter pools. Or put another way, the partitioning of C into various pools of differing N content must reproduce the measured C:N of the soil as a whole. Prior to clearing, large amounts of fresh organic matter would have been present in surface soil within the native plant community. We set the C:N of fresh organic matter to a value of 8 based on the ratio of losses of C and N from the profile and have assumed that much of this material will be lignin from partially decomposed organic matter in a native leguminous plant community at a climax state. The C:N of the humic pool is taken from step (i) above. The parameterisation derived from this logic is shown in Table 1. This configuration should provide a linked decline in soil C and N, and thus the observed change in bulk soil C:N over time as the faster, low C:N pools decline.

Table 1
Initial C and N pools (kg ha^{-1}) for layers in the surface 0.3 m.

	Depth (m)			Total
	0–0.1	0.1–0.2	0.2–0.3	
Carbon (kg ha^{-1})				
Inert	7590	2460	2600	12650
Humic	17348	10933	9556	37837
Microbial	867	546	477	1890
FOM	11600	94	92	11786
Total	37405	14033	12725	64163
Nitrogen (kg ha^{-1})				
Humic	1355	854	746	2955
Microbial	108	68	59	235
FOM	1450	2	2	1454
Total	2913	924	807	4644

The final data required for model initialisation concerns the input of C and N to the soil surface after clearing of the native vegetation. Bulldozers were used to fell trees and these were left on the ground for many months before burning, raking of unburnt coarse woody debris, and cultivation (Cowie et al., 2007). Surface litter and felled foliage, bark and twigs from standing trees would have been susceptible to decomposition in the period before burning. Coarse woody debris would not have decomposed significantly in the period before it was burned and removed. Measurements of total C and N content of surface litter and standing vegetation have been made for Brigalow forests (Moore et al., 1967; Dowling et al., 1986). From this data we estimate that approximately 40 t ha^{-1} of biomass with a C:N ratio of approximately 30 would have been on the soil surface subsequent to clearing.

2.2. Results and discussion

APSIM was able to adequately describe the major processes and resultant changes in soil C and N content within the surface (0–0.3 m) soil layers. The observed and predicted time courses of crop productivity and soil fertility are demonstrated in Fig. 2. Clearing of native vegetation resulted in a rapid decline in soil C and N in the surface 10 cm and these trends are captured by the model (Fig. 2a,b). The model was also able to predict the levels of C and N within the soil once the rapid losses of labile material had been completed and the input and decomposition of soil organic matter approached a new equilibrium. Predictions of general productivity are similar to observation though the yields in some seasons showed large discrepancies (Fig. 2c). Some of these can be attributed to the impacts of weeds, pests and diseases which are not represented within the model. It is therefore not unexpected that the model would significantly over-predict measurement in some seasons. The impact of declining fertility upon productivity has previously been illustrated using the observed time trends in protein content of wheat grain (Radford et al., 2007). A very similar trend can be observed in the measured and predicted wheat grain protein contents (Fig. 2d). This gives confidence that impact of declining fertility upon crop growth is being captured by the dynamic model.

Although neither denitrification nor N_2O emissions were measured as part of this long term study, confidence in the model predictions of these processes can be obtained in two ways. The first is by comparing the model predictions of the main drivers of denitrification such as C, water and N. If these processes are

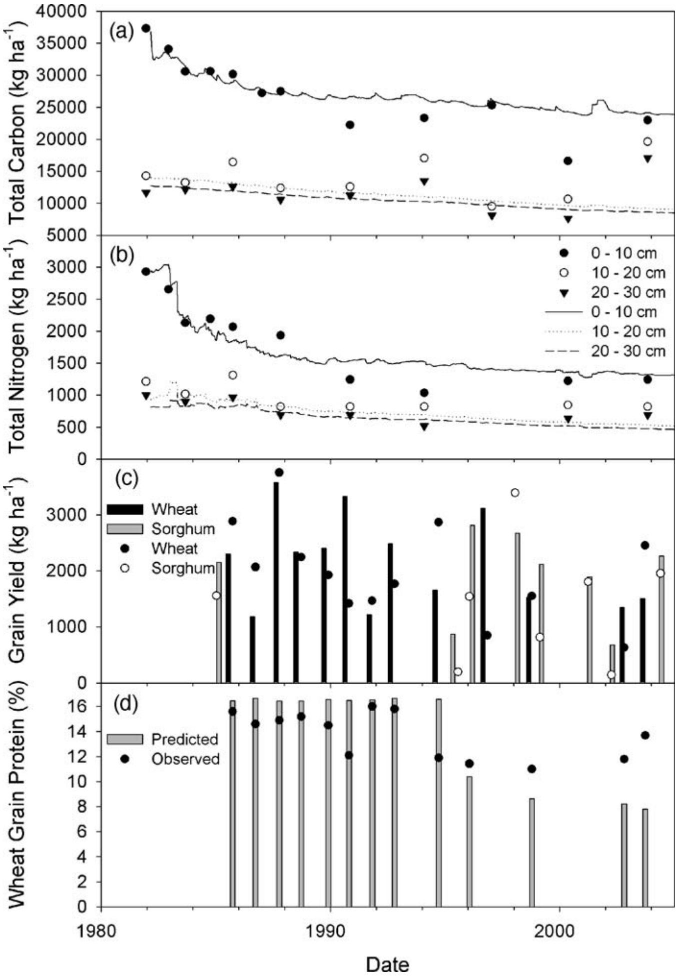

Fig. 2. Observed and predicted time courses of (a) soil C and (b) total soil N (observed as symbols and predicted as lines, see legend) as well as (c) grain yield and (d) wheat grain protein content (observed as symbols, predicted as bars).

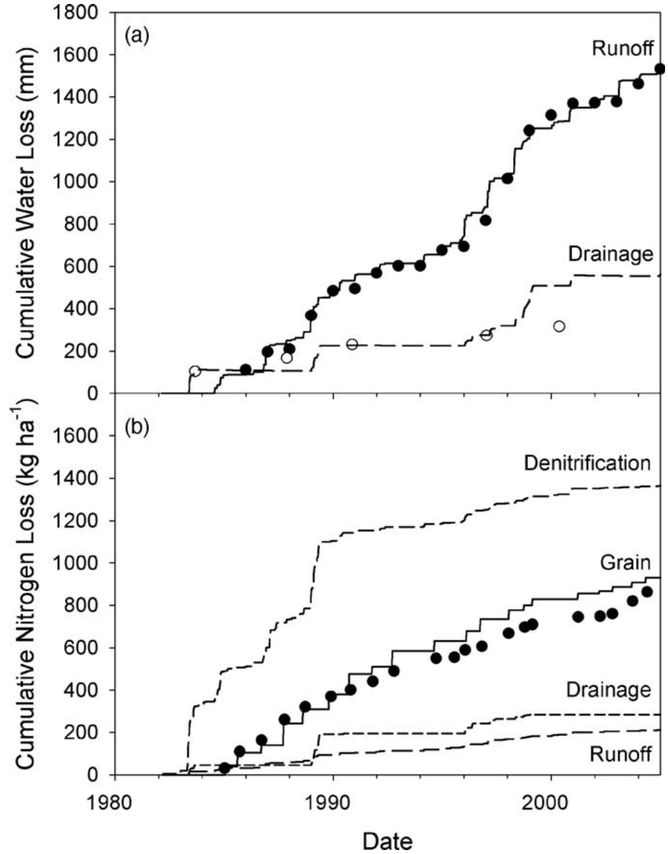

Fig. 3. Observed and predicted accumulated losses for (a) water (runoff and drainage) and (b) N (export in grain, runoff water or drainage and denitrification). In both figures observations are as symbols and predictions as lines.

captured adequately, predictions of denitrification are likely to respond in an appropriate way. The second approach is via study of the overall N balance and its components.

In terms of the drivers of denitrification, we have already demonstrated that the model is predicting the observed changes in C and N content within the soil. Fig. 3a shows that the water balance was similarly well simulated. Cumulative runoff and drainage losses of water follow measured values (Thornton et al., 2007; Silburn et al., 2009). Predictions of total soil moisture storage to 2 m were also compared to measurements taken with a neutron moisture probe (Fig. 4). Predictions of chloride displacement within the soil profile were compared with data from analyses of deep soil cores (data not shown). These results provide confidence in predictions of the C, N and water processes driving denitrification.

We can now gain confidence in the magnitude of the simulated denitrification losses by accounting for each of the other N losses as part of the overall balance in which nearly 2400 kg N ha^{-1} was lost from the surface soil in the first 9 years after clearing, an average of 266 kg N ha^{-1} year^{-1}. Losses from the system include export in harvested grain and losses in deep drainage and runoff. Fig. 3b illustrates the predictions of each of these terms. Predictions of cumulative N export via harvested grain are close to measured values with a time trend that reflects the changing rate of fertility over time. Nitrate leaching data is not available but there is evidence that model predictions should be accurate. Firstly, the good predictions of deep drainage (Fig. 3a) and depth of chloride leaching (data not shown) indicate the simulated values should be reasonable. Measurements of deep soil nitrate are limited but show no sign of N accumulation at depth. Moreover, it is unlikely that leaching should account for a large amount of the overall loss given that much of the decrease in total soil N occurred during long periods of low or negligible drainage indicated by the time series of measured soil chloride profiles (Silburn et al., 2009) and the current model. In this same period, observed and predicted soil water content rarely exceeded the calculated water holding capacity of the soil (Fig. 4) indicating low levels of drainage. Predictions of N lost in runoff were calculated using predicted

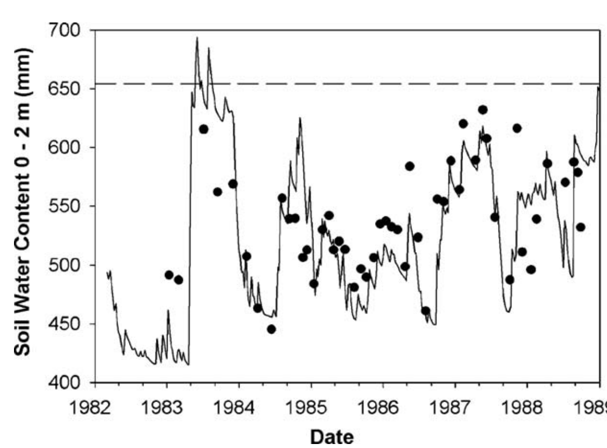

Fig. 4. Measured (symbols) and predicted (solid line) soil water content to 2 m depth. Dashed line indicates the soil water content at the drained upper limit (see Fig. 1).

runoff volume and the average N concentration of runoff water measured during a 5-year time period within the study. Assuming that these N concentrations did not differ significantly outside the measurement period, this loss mechanism should be well captured by the model given the adequacy of the runoff predictions (Fig. 3a). The only remaining major N loss mechanism is denitrification. Whilst not measured itself, we conclude that the predictions are robust given the good prediction of the overall N balance and the other major loss processes.

The predicted time trend in denitrification losses consists of two distinct phases, as for changes in soil C and N. In the early years after clearing (1982–1990), soil C and N levels were high, rainfall was high in several years, and the soil remained fallow for long periods of time due to a low cropping frequency (one winter crop per year) adopted in this phase of the BCS (Cowie et al., 2007). For this early period (1982–1990), the predicted average annual denitrification was approximately 127 kg N ha^{-1} $year^{-1}$. While at first this denitrification rate appears high for an unfertilised rain fed system, more closely resembling values found in high input irrigated fields (Rochester, 2003; Scheer et al., 2008), it is plausible given the highly fertile soil and extremely wet summer fallows during this early period. Denitrification losses during the later years were significantly lower with an average of only 15 kg N ha^{-1} $year^{-1}$ being lost from 1990 to 2005. These simulated losses compare with denitrification rates of 31 kg N ha^{-1} $year^{-1}$ measured over 2 years under cropping in another Brigalow soil that have been cropped for more than 10 years (Weier et al., 1991). These lower losses reflect the reduced soil C and N levels, the dry climate of the early 1990s, and an increased cropping frequency (the incorporation of summer crops into the rotation) adopted during this phase of the BCS (Cowie et al., 2007). Although untested in this region, predicted N_2:N_2O ratios for the scenarios compared well to those measured on agricultural soils which were previously under Brigalow forests (Weier et al., 1993).

The results from this testing on the BCS are encouraging and indicate that the model configuration is able to adequately capture the main elements of the C, N and water balances of these farming systems and is therefore suitable for the proposed scenario analysis.

3. Model application

3.1. Method

The model configured and tested above was used to estimate the changes in soil C and N_2O emissions for the following cropping systems:

(1) WS – wheat and sorghum with no fertiliser;
(2) WS + N – wheat and sorghum with fertiliser;
(3) WS + N split – wheat and sorghum with a split application of fertiliser;
(4) CS – chickpea and sorghum, sorghum fertilised;
(5) WM – wheat and mungbean, wheat fertilised.

All agronomy was specified according to local best practice. Fertiliser requirements were calculated at sowing so as to raise the soil mineral N within the crop root zone to 100 kg N ha^{-1}. The third scenario provided a split application of fertiliser to investigate the benefits of better matching N supply to crop N demand. In this case, the calculated fertiliser requirement was split with 30% applied at sowing and the remainder at 30 days after sowing for sorghum and 40 days after sowing for wheat. These times correspond roughly to the beginning of stem elongation. Initial soil C levels were set to those simulated for 2005 during the model testing exercise as these represent the soil organic C status of much of the farming land within the region that has been cropped for many decades since the Brigalow forest was cleared. Simulations were undertaken using weather data from 1950 to 2005 to provide a longer sample of local climatic conditions. Partial accounting for greenhouse gas emissions and economic return was applied to provide a simple comparison of the relative performance of the scenarios. Changes in soil C content (0–0.3 m) were used to calculate average annual carbon dioxide (CO_2) emission from soils. Predicted N_2O emissions, expressed as CO_2 equivalent, were combined with this to provide a value for total emissions for soils. A partial gross margin for each scenario was calculated using predicted yields and cropping history and representative costs and prices published by the Queensland Department of Primary Industries (www.dpi.qld.gov.au). These include machinery, seed, fertiliser, herbicide, pesticide and harvesting costs. Owner or labour costs are excluded.

3.2. Results and discussion

Predicted long term N inputs, economic return and soil greenhouse gas emissions are detailed in Table 2. For the unfertilised WS scenario, soil C continued to decline through the period from 1950 to 2005. This result is similar to measured trends for surface soils in a chronosequence study for similar soils within south-eastern Queensland (Dalal and Mayer, 1986). N_2O losses remained relatively low reflecting reduced soil N and C. The addition of N to the system via fertiliser increased crop growth and so slowed the run down of soil C and halved CO_2 emission. However, it substantially increased the amount of N_2O produced. The increases in N_2O emissions above those for the unfertilised scenario represent approximately 2.2% of the fertiliser N applied for WS + N and 1.9% for WS + N split. These emissions relative to fertiliser N application are slightly higher than estimates employed for inventory reporting purposes (IPCC, 2000) and much higher than measured for rain fed fertilised crops on sandy soils in semi-arid Australia (Barton et al., 2008). However, they are less surprising considering previous measurements from Brigalow clay soil (Weier et al., 1991) and the soils, climate and cropping frequency in this region. These results support previous studies which indicate that improving the method of fertiliser application may provide an effective way to reduce emissions (Del Grosso et al., 2009). Studies would be required to look into the feasibility of approaches such as split application, or alternative fertiliser treatments, for improving the timing between crop N supply and demand within these environments.

The impact of incorporating legumes into cereal rotations was not consistent. The WM rotation was as effective as the fertilised cereal rotation in maintaining top soil C levels but had lower N_2O emission resulting in a total emission similar to the zero input (WS) scenario. In contrast, the CS rotation resulted in declining soil C and relatively high N_2O emission. Investigation of the CS simulations showed that chickpea crops had both direct and indirect impacts on soil C input. Chickpea biomass production is lower than wheat and therefore so is its C input. As a further consequence, chickpea provides very little protection to the soil surface via stubble cover going into the summer when temperatures and rainfall increase. A reduction in soil water storage due to increased evaporation and runoff causes delayed or reduced sorghum sowing opportunities. This delay increased the duration of summer fallow periods in the simulations during which N_2O and CO_2 emissions are likely to be highest. On the other hand, the wheat crops in the WM scenario provided high levels of stubble and soil cover during the summers increasing soil water storage and opportunities for sowing. The benefits of maintaining stubble cover been shown to be of considerable importance for cereal cropping in the northern grain growing regions of Australia (Thomas et al., 2007).

Table 2
Simulated (1950–2005) average production, fertiliser N input, CO_2 and N_2O emissions and economic return for the 5 management scenarios. Scenarios are described within the text.

	System				
	WS	WS + N	WS + N split	CS	WM
Average grain yield (kg ha^{-1}) (number of crops)					
Winter crop	847 (48)	2484 (34)	2552 (32)	1805 (32)	2564 (33)
Summer crop	1460 (56)	3539 (51)	3553 (51)	3203 (42)	1438 (55)
Average annual N application (kg ha^{-1} year^{-1})					
Winter crop	0.0	32.6	31.1	0.0	25.6
Summer crop	0.0	55.1	52.6	36.4	0.0
Total	0.0	87.7	83.7	36.4	25.6
Average annual emissions (t CO_2e ha^{-1} year^{-1})					
Carbon dioxide (CO_2)	0.96	0.48	0.50	0.63	0.49
Nitrous oxide (N_2O)	0.27	1.20	1.00	1.03	0.70
Total	1.23	1.68	1.50	1.67	1.20
Average annual gross margin ($ AUD ha^{-1}year^{-1})					
Winter crop	19.5	237.2	236.1	331.2	251.3
Summer crop	−15.0	304.3	310.9	221.5	526.4
Total	4.5	541.5	547.0	552.7	777.7
Economic return on emission ($ AUD t CO_2e^{-1})	3.6	323.2	364.2	331.3	650.2

Fig. 5 demonstrates the distribution of N_2O emissions across the year for each of the scenarios. In all systems, the previously described seasonal trends in N_2O emissions on Brigalow soils in response to temperature (Weier et al., 1991) and C and N supply (Weier et al., 1993) are evident. The overall impact on N_2O emissions of the different cropping systems is varied when compared to a standard WS + N rotation. N_2O emissions during summer and winter planting windows, are reduced by split application of N (WS + N split) due to the reduced amounts of soil nitrate during crop establishment. For WM there is a slight increase in N_2O emissions in winter as mungbean residues mineralise through autumn. However, there is a large decrease in N_2O emissions in summer when N fertiliser use is no longer required. The CS rotation shows a slight decrease in N_2O emissions in winter when N fertiliser is no longer needed but a large increase in summer when decomposition of chickpea residues increases denitrification (Fig. 5). The vast difference in these scenarios illustrates the need to consider the complex feedbacks and agronomic impacts of various greenhouse gas intervention strategies. Approaches to decrease emissions, such as incorporating legumes, will need to be tailored to individual cropping systems or the outcomes may be counter productive.

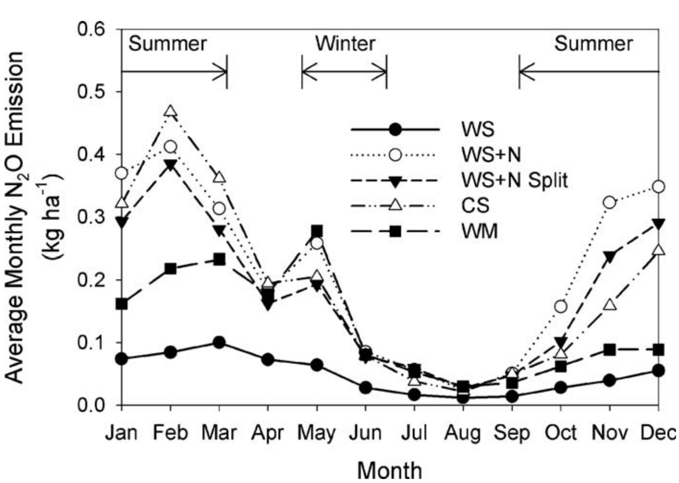

Fig. 5. Predicted changes in seasonal N_2O emissions due to cropping/fertiliser management. Approximate winter and summer planting windows are shown in order to indicate likely periods of fertiliser application.

The economic and agronomic performances of the scenarios show the expected trends between high and low input production systems. The traditional low, or no, input scenario is no longer profitable for soils that have been cropped for several decades due to the rundown in soil fertility. Increased N supply through the use of fertilisers or legume crops significantly increased predicted production and gross margins, especially when a high value crop such as mungbean is incorporated into the crop rotation. N inputs for cereal crops were decreased by up to 34% when grown in rotation with legumes. Split application of fertiliser had only a minor impact on crop production but decreased average annual greenhouse gas emission by 10%.

The ratio of annual gross margin to annual emission provides an indication of the trade off between production and environmental goals of each scenario. All high input scenarios provided a higher return per unit emission when compared to the zero input scenario. The return on emission for the WS + N, WS + N split and CS rotations was similar due to similarity in both the economic and emission components. The return on emission for the WM scenario was nearly twice that of the other high input systems due to both reduced greenhouse gas production and increased economic return. Whilst this value is sensitive to price fluctuations for the high value crop (mungbeans) a reduction in gross income of 42% would be required to reduce returns per emission back to the other high input scenarios.

4. General discussion

Our study highlights some important issues in managing CO_2 and N_2O emissions together in farming systems. We predict that N_2O emissions from rainfed grain-based farming systems in the Australian subtropics are likely to be higher than anticipated from experience in similar farming systems in temperate areas. While this conclusion is consistent with the limited experimental data available to date (Weier et al., 1991, 1993), it requires broader experimental confirmation.

We also predict that there are crop rotation variations within farming systems that can help reduce these emissions. However, the impact of management change within rotations is complex: It would have been difficult to deduce the direction or magnitude of changes in CO_2 or N_2O emissions in the different systems simulated *a priori* in such an environment. In the case of including legume in a crop rotation to reduce N fertiliser needs, there are impacts upon hydrology, cropping frequency and time of sowing; any of these may in itself have as big an impact as the benefit expected from N fixation. These complex interactions between crop management and C, N and water balances demonstrated in our analyses add weight to the argument that process-based models have much to contribute in the study and quantification of emissions from agricultural systems where climate, soils and management combine in complex ways (Del Grosso et al., 2009).

This paper also highlights the issue of parameterisation of soil process models. For example, whereas others have explored the use of targeted laboratory techniques to parameterise individual soil C pools (Skjemstad et al., 2004), we have demonstrated that simple logic applied to emergent soil behaviour can be used in a similar way. The choice between these two approaches to parameterising soil C pools is similar to that available in soil hydrology, where soil hydraulic properties can be obtained from laboratory or functional measures. Both approaches are valid (Williams et al., 1991). The differing rates of decay of C and N, the resultant changes in soil C:N ratio, and the decline in crop productivity are all well understood for these systems and so model parameterisation must take all these factors into account in a simple yet meaningful way. For example, failure to account for the changes in N mineralisation with changing soil C composition will impact upon predictions of crop production, which is the source of C input into the soil. Modelling of soil C and N should combine attempts to measure soil C composition with considerations of soil function and the method employed above provides a framework for such considerations.

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