Can the sequestered carbon in agricultural soils be maintained under changes in management, temperature and rainfall? A sensitivity assessment

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Abstract

Carbon (C) sequestration in agricultural soils has the potential to mitigate climate change and helps sustain soil productivity. Usually, continual nutrient input and residue retention are required in order to reach the C sequestration potential and maintain the sequestered C. However, few studies have assessed the vulnerability of the sequestered soil C to changes in agricultural management and climate. Here we applied the Agricultural Production Systems sIMulator (APSIM) to simulate the soil C dynamics to equilibrium under optimal management with 100% residue retention and no nitrogen (N) deficiency at 613 sites across the Australian croplands. We examined the response of sequestered soil C to potential warming and rainfall change, as well as suboptimal management with reduced residue retention and/or N input. On average, soil C was lost at rate of 0.14 Mg C ha⁻¹ yr⁻¹ when residue retention was reduced by half. Removing all residues doubled the rate of C loss (i.e., 0.28 Mg C ha⁻¹ yr⁻¹). Reducing application rate by half of the optimal N rate or to zero led to C loss of 0.089 and 0.27 Mg C ha⁻¹ yr⁻¹, respectively. Multivariate linear regression analysis indicated that C loss rate increased with active C stock (non-inert C) in the sequestered C. Given an active C stock, the loss rate increased with increasing temperature and/or rainfall. Future warming was estimated to increase soil C loss, especially in cooler and/or wetter regions. The effect of potential rainfall change was relatively moderate and depended on the

direction (increase or decrease) of rainfall change. Management strategies for effective C sequestration in agroecosystems should and can be developed based on local climatic conditions and soil-specific amount of active organic C.

Keywords: carbon loss, carbon input, carbon sequestration, climate change, management practices, sequestration potential, vulnerability

1. Introduction

Carbon (C) sequestration in agricultural soils has been recognized as an effective approach to mitigate climate change, if management practices can be improved, such as adoption of no-tillage, residue retention and fertilizer application matching crop nutrient demand (Freibauer et al., 2004; Lal, 2004; Post and Kwon, 2000; Smith, 2004). The role of no-tillage in soil C sequestration has been questioned by recent studies (Baker et al., 2007; Luo et al., 2010a; Powlson et al., 2014). These studies suggested that different tillage practices just altered C distribution with depth but not the total amount of soil C. Residue retention and fertilizer application, however, are widely accepted for enabling C sequestration (Dalal et al., 2011; Khan et al., 2007; Liu et al., 2014; Luo et al., 2014; Luo et al., 2013; Zhao et al., 2013) because of their direct effect on C inputs. Wang et al. (2015) estimated that an average C input of $2.1 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$ is required in order to stop soil C loss in China's croplands and $5.1 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$ to approach the global mean of 55 Mg C ha^{-1} in the top 30 cm soil by 2050. The long-term experimental results in the United States also showed that soil C sequestration rate was significantly and positively correlated with C input (Kong et al., 2005), consistent with other findings in Europe (Smith et al., 1997) and Australia (Luo et al., 2014; Zhao et al., 2015). All these findings suggest that continual input of C is critical to increase or maintain soil C under a specific cropping system, soil and climate.

The amount of C input, however, is very sensitive to changes in management, such as

residue retention and fertilizer application, [due to their impact on](#) residue production. Optimal management (e.g., 100% residue retention and/or no nutrient deficiency) is usually assumed to estimate soil C sequestration potential (Freibauer et al., 2004; Luo et al., 2013; Smith, 2004; Yagasaki and Shirato, 2014), [although](#) the optimal management for C sequestration may not necessarily [be](#) optimal for other [outcomes such as economic return, resource use efficiency, and environmental impact](#). For example, excessive nitrogen (N) fertilizer application can pollute and degrade the environment (Khan et al., 2007; Powlson et al., 2011) as evidenced by the eutrophication of water (Moss, 2008), greenhouse gas emissions (Zaehle et al., 2011), and soil acidification (Guo et al., 2010). Residue retention may be also limited by residue harvest for bioenergy (Elshout et al., 2015; Zhao et al., 2015) and feed production. The potential changes in these management practices will [influence both](#) C input [and](#) soil environment [such as](#) microbial activity, soil water retention and nutrient availability. The consequences of management change on the sequestered C [need to be quantified in order](#) to identify context-specific effective management strategies for persistent C sequestration in agricultural soils under diverse management and environmental conditions across large scales.

Climate change ([e.g., warming and changes in rainfall](#)) is another factor that influences the fate of sequestered C through its effect on C input and decomposition. Previous studies mainly focused on the effect of climate change on the estimation of soil C sequestration potential (Álvaro-Fuentes et al., 2012; Grace et al., 2006; Jiang et al., 2014; Thomson et al., 2006) but not the fate of the sequestered C. The sensitivity of sequestered C to climate change may vary with local soil and climatic conditions. It will impact the accounting of C in agro-ecosystems and assessments of the role of agricultural soils in climate change mitigation.

Long-term soil C change induced by management and climate changes ([e.g., warming and rainfall change](#)) is difficult to quantify based on observational approaches and datasets (e.g., soil inventories and field experiments). For example, it is impractical, if not impossible,

to implement field experiments with multiple agricultural management practices over large spatiotemporal scales. Although soil inventories and baseline datasets can provide reliable and spatially explicit data on soil C (Viscarra Rossel et al., 2014), alone they cannot be used to detect long-term changes in soil C or to understand the corresponding underlying mechanisms causing such changes (Meersmans et al., 2008; Meersmans et al., 2011). Process-based models can be used to simulate the biogeochemical processes influencing changes in soil C over large space and long periods of time. Indeed, a number of models including CENTURY (Parton et al., 1987), RothC (Jenkinson, 1990) and Agricultural Production Systems sIMulator (APSIM) (Keating et al., 2003) have the capability of modelling the effects of various management and environmental interventions, and thus have been widely used to simulate soil C dynamics in agro-ecosystems. The process-based models have the advantage that they can be used to explicitly analyse the impacts of management practices and environmental variables on soil C dynamics. In addition, simulation outputs of process-based models can be summarised to understand the primary factors that affect soil C dynamics (Luo et al., 2013; Marie and Simioni, 2014).

In this study, we assessed the vulnerability of soil C that was sequestered under the optimal management (100% residue retention and no fertilizer deficiency) to changes in management options with reduction in C input, potential warming and rainfall change. Using the APSIM model and detailed soil profile datasets at 613 sites across Australian cereal growing regions, we simulated soil C dynamics in a typical continuous wheat system (growing wheat every year) under different scenarios of management, temperature and rainfall changes. Our objectives were to i) quantify the vulnerability of sequestered C (expressed as C loss) to potential changes in management, temperature and rainfall, ii) assess the spatial pattern of C loss, and iii) investigate how the C loss correlates to climatic and soil conditions.

2. Materials and Methods

2.1 Data sources

The study region covers the whole Australian cereal growing regions (Fig. 1). Within this region, there are 613 soil sites where detailed soil profile data are available. The soil sites are roughly randomly distributed in the Australian grain regions (Fig. 1). These are point soil profile data collected by Agricultural Production Systems Research Unit, available via the Australian Soil Resource Information Systems (<http://www.asris.csiro.au/mapping/hyperdocs/APSRU/>). They are fully characterized soil profiles with information needed to run APSIM model, including soil bulk density, organic C content, hydraulic properties (saturation water content (SAT), drained upper limit (DUL), 15 bar lower limit (LL15)), pH etc for each soil layer. Daily recorded weather data from 1957 to 2010, including daily global radiation, rainfall, maximum and minimum temperatures, are available from the Australian Bureau of Meteorology weather stations. These data are obtained from SILO Patched Point Dataset (<https://www.longpaddock.qld.gov.au/silo/>).

2.2 The APSIM model

APSIM simulates biophysical processes in farming systems, and has been comprehensively verified and used to study productivity, nutrient cycling and environmental impacts of farming systems as influenced by climate variability and management interventions (Holzworth et al., 2014; Keating et al., 2003). APSIM simulates crop growth and soil processes on a daily time-step in response to climate (i.e., temperature, rainfall, and radiation), soil water availability, and soil nutrient status (i.e., N and Phosphorus) (Wang et al., 2003).

APSIM simulates the dynamics of both C and N in the soil. Soil organic matter is divided into four conceptual pools (i.e., fresh organic matter, microbial biomass, humic and inert organic matter) and decomposition of each pool is treated as a first-order decay process,

leading to release of CO₂ to the atmosphere and transfer of the remaining decomposed C to other pools. For each pool, a maximum potential decomposition rate constant is assumed, and the actual decomposition rate is obtained by modifying the maximum rate according to soil moisture, temperature and nutrient availability. Flow of N depends on C:N ratio of the receiving pool. Decomposition of surface residue also takes into account of the degree of contact of residue with soil to modify the maximum decomposition rate of residue. APSIM allows flexible specification of management options such as crop type, residue management, and fertilizer application.

APSIM has been tested for its performance to simulate soil organic C dynamics against measurement data in a number of studies (Huth et al., 2010; Luo et al., 2011; Probert et al., 2005; Thorburn et al., 2001; Wang et al., 2013). Results generally showed that APSIM is able to predict soil organic C dynamics under different cropping and management systems at many locations and over long time periods. This forms the basis for us to use the model in the current study.

2.3 Model simulations and scenario analysis

Due to the variation in SOC initial values and the different trajectories of soil organic C change caused by complex interaction of climate, soil and management conditions, the time needed for simulated SOC to reach the equilibrium state may be as long as 900 years (Luo et al., 2013). Therefore, 1020 years of climate data series were generated by randomly sampling the historical data from 1957 to 2012 with replacement. This treatment of data extension does not include any climate change trend.

The APSIM model was first run for 1000 years at the 613 references sites in order to reach the equilibrium state of soil organic C dynamics. A continuous wheat system was assumed for the simulation under 100% residue retention. To identify the optimal N requirement for C sequestration, the simulation was run under an N fertilizer application

scenario of 0–300 kg N ha⁻¹ yr⁻¹ in 20 kg N ha⁻¹ increments. In all except the non-fertilizer application scenario, 25 kg N ha⁻¹ was applied as a base fertilizer incorporated into soil at time of sowing, with any additional N being applied at the stem elongation stage as a top-dressing (broadcast). Under each of the 16 N application rates and each site, we regarded the average of crop yield and soil organic C stock (in the top 30 cm soil) in the last 100-year simulations as the crop yield and soil C stock at the equilibrium state respectively.

At a **specific** site, crop residue production is the predominant factor controlling soil C **input** in the simulation. In real croplands, however, crop production rarely reaches the maximum potential because of the **declining of economic return under very high N rate** (Carberry et al., 2013). In addition, we did not modelled other production constraints such as pests, diseases and/or trace element deficiencies. Thus, we treated the N application rate obtained the 90% of the simulated maximum crop yield as the optimal N rate for C sequestration. The soil C stock under this **optimal** N rate was regarded as the maximum potential of soil C sequestration.

After the equilibrium state **of soil C was reached**, the model was run for **further 20** years to assess the vulnerability of sequestered soil organic C under different management, **temperature and rainfall** change scenarios. **All data analysis in this study was based on results during 20 years simulations post-equilibrium state**. Two management scenarios were selected: i) four levels of reduction of N fertilizer application (25%, 50%, 75% and 100% **of optimal**), and ii) four levels of residue removal (25%, 50%, 75% and 100%).

To quantify the effects of **potential warming and rainfall change** on the sequestered soil C, a sensitivity test was performed with temperature increases by 0.5, 1, 1.5 and 2 °C **and** rainfall change of –20%, -10%, +10% and +20%. The simulation under the identified optimal management practices based on historical climate data was treated as baseline.

2.4 Attributes regulating the vulnerability of sequestered C

To assess the vulnerability of sequestered soil C, we calculated, for each management, temperature and rainfall change scenario at each site, the annual soil C loss rate (C_{loss}) in the top 30 cm soil ($\text{Mg C ha}^{-1} \text{ yr}^{-1}$) over the post-equilibrium 20-year simulations. The C_{loss} was calculated as the amount of soil C loss (the difference between soil C stock under optimal management and baseline climate and that under those scenarios) divided by the years of the simulation (i.e., 20 years). This approach defines a higher vulnerability with a greater C_{loss} . The distribution and spatial pattern of C_{loss} at the 613 sites were compared among the management, temperature and rainfall change scenarios.

After deriving the vulnerability expressed as the loss rate C_{loss} , we further asked the following question: how and why does C_{loss} change across different locations? We hypothesized that C_{loss} was associated with active soil organic C (non-inert C) and climatic conditions including temperature and rainfall. Active soil organic C indicates the total amount of soil C that potentially subject to decomposition, and was calculated as the difference between total soil C at the equilibrium state and amount of inert organic C which was specified in the initial soil profile dataset for each soil. The two climatic factors represent the dominant factors regulating decomposition, and were calculated as the average for the post-equilibrium 20-years. Rather than fitting a multivariate regression model complicating underlying interpretation, we fitted following model for each scenario to assess the effect of active organic C (C_{active}) and regulation of temperature (T) and rainfall (R) on this effect:

$$C_{loss} = C_{active} (\alpha + \beta R + \gamma T), \quad (1)$$

where R and T are the standardised (z -score) mean annual rainfall and temperature, i.e., subtracting rainfall (or temperature) at a given site by the mean in the 613 sites and dividing by 2 standard deviations (SD); α is the predicted C_{loss} (i.e., the slope) corresponding to a 1 t ha^{-1} change in C_{active} when R and T are equal to zero, i.e., rainfall and temperature are the

same as the average of the 613 sites; β is the slope relative to rainfall, corresponding to the slope change per 2 *SD* mm change in rainfall; γ is the slope relative to temperature, corresponding to the slope change per 2 *SD* °C change in temperature. Across the studied 613 sites, the mean and SD of temperature and rainfall are 17.5 and 1.8°C, 460 and 128 mm, respectively. For application of this model at specific location, local temperature and rainfall should be scaled accordingly. The coefficients estimated based on *z-scores* have the advantage of indicating the relative importance of rainfall and temperature: the greater the absolute value of the coefficient, the more important the rainfall or temperature.

This model assumed that the vulnerability directly related to the amount of C_{active} (coefficient α), which was modified by local rainfall and temperature (coefficients β and γ). We did not fit an intercept for the model as when C_{active} was equal to 0, the loss rate C_{loss} must be also 0. The coefficients were compared between the management scenarios. If α is positive, C_{loss} will increase with C_{active} , and *vice versa*. If β is positive, given C_{active} and temperature, C_{loss} will increase with rainfall, and *vice versa*. If γ is positive, given C_{active} and rainfall, C_{loss} will increase with temperature, and *vice versa*.

3. Results

At the equilibrium state under optimal management, average soil C reached to 52.4 t ha⁻¹, corresponding to an average increase of 9.3 t ha⁻¹ (Fig. 2), and the vulnerability of this sequestered soil organic C (calculated as the soil C loss rate C_{loss}) to management and climate changes are presented in Fig. 3. On average, C_{loss} linearly increased with reduction of N fertilizer application and residue removal fraction. Temperature increase positively related to C_{loss} , but the effects of rainfall change, on average, were relatively negligible. Across the full set of simulations of the 613 sites, variation in the vulnerability was relatively large, and increased with fertilizer application reduction and residue removal level and temperature increase (Figs. 3a, b and c). Fig. 4 shows the spatial pattern of C_{loss} under four case scenarios

for 613 sites across the study area. The spatial pattern was very similar between the scenarios of 100% fertilizer reduction, 100% residue removal and 2 °C temperature increase. In general, C_{loss} decreased from southeast to northwest in the study area other than Western Australian. Under 20% rainfall decrease, soil C loss mainly occurred in Queensland and the west of South Australia. In other regions, soil C sequestration was enhanced (i.e., negative C_{loss}), particularly in the east of South Australian and west of Victoria.

The large spatial variability of the vulnerability could be explained by the amount of active organic C (C_{active}) and its interaction with local temperature and rainfall conditions (Fig. 5). Under fertilizer application reductions, residue removals and temperature increases, C_{loss} increased with C_{active} . Given a typical C_{active} , C_{loss} were higher in higher temperature and rainfall regions under fertilizer reduction and residue removal scenarios, and in lower temperature and higher rainfall regions under temperature increase scenarios. C_{active} and its interaction with temperature and rainfall explained more than 70% (ranging from 73%–96%) of the variation in C_{loss} across the study area (Fig. 5d). Fig. 6 highlights the relationship between C_{loss} and C_{active} as impacted by background temperature and rainfall under the case scenario of 100% residue removal. With decreases in rainfall, C sequestration increased with C_{active} (negative coefficient α). However, this increase was limited by higher background temperature and rainfall conditions. With increases in rainfall, C_{active} , temperature and rainfall showed the opposite effect to that under rainfall decrease (Fig. 5).

4. Discussion

The simulation results in this study demonstrated that sequestered soil organic C through adopting optimal agricultural management could be rapidly lost if halting the optimal management. Sequestered soil organic C responded differently to potential warming and rainfall change. On average, C loss rate increased with reduced fertilizer applications, removal of residues and temperature increase. This highlights the critical role of C input and

temperature in determining the amount of organic C in soil (Davidson and Janssens, 2006; Wang et al., 2015). The effect of rainfall change was moderate and uncertain depending on the direction (increase or decrease) of the rainfall change. Spatially, vulnerability of the sequestered C to changes in management and climate varied widely depending on site-specific conditions. Site-specific amount of active organic C, local rainfall and temperature explained most of the variability between locations. The identified relationship (Eqn. 1 and Fig. 5) can be used as space-for-time substitution to infer trajectories of post-equilibrium soil C dynamics under different soil C level and climatic conditions (Luo et al., 2013; Marie and Simioni, 2014).

The vulnerability of sequestered soil C to change in management practices positively correlated with the amount of active organic C stock (Fig. 5 and Fig. 6), i.e., the greater the active organic C stock, the larger the loss rate. This finding is consistent with those from other studies (Bellamy et al., 2005; Zhao et al., 2013). Here, we defined the active organic C as the difference between total soil C and inert organic C, and inert organic C is the C component considered to be recalcitrant to decomposition because of physicochemical and biological protections. Our results suggests that any changes in soil environment (e.g., microbial activity, soil aggregate and soil mineralogy) that regulate C accessibility and decomposability may influence the vulnerability of sequestered soil C (Jastrow et al., 2007). Under the same level of active organic C, however, C loss rate was larger under higher rainfall and/or temperature conditions, mainly resulting from the relevant faster decomposition process in wetter and/or warmer regions (Carvalhais et al., 2014). Thus, sequestered soil C should be more resistant to management changes in cooler and/or drier regions. Except the significant effects of both rainfall and temperature, it also should be noted that rainfall is much more important than temperature to regulate the loss rate as evidenced by the larger absolute value of the coefficient for rainfall in Eqn. 1 (Fig. 5). This suggests that

rainfall condition should be the prior consideration when developing site-specific management strategies for persistent C sequestration.

The response of sequestered C to future climate change affects our assessments of the feasibility of long-term C sequestration in agricultural soil and its capacity to mitigate climate change. Our simulations indicated that warming and rainfall change had distinct effects on sequestered soil C. Warming generally reduced the sequestered soil C, particularly in cooler and/or wetter regions. In these regions, higher temperature will not only stimulate soil C decomposition but also limit crop growth (Asseng et al., 2015; Davidson and Janssens, 2006). Both processes have negative effect on soil C sequestration. Rainfall change usually has opposite effects on soil C decomposition and C input (from crop residue production), resulting in either positive or negative effect. The large variability caused by changes in rainfall suggests that there is potential for further enhancing organic C sequestration under future changes in rainfall, particularly in cooler and/or drier regions. Further study on the effects of the interaction between temperature and rainfall changes and baseline climate condition on the sequestered soil C is needed.

There are several uncertainties and limitations in the simulations of this study, warranting further study. First, the vulnerability of sequestered soil C would vary with other management changes other than residue and fertilizer management (Lugato et al., 2014; Luo et al., 2010b). For example, a simplified continuous wheat system was assumed for the simulation. Although wheat production is common in the region, one limitation of this assumption is that it does not capture the effect of different quality of C inputs (e.g., C:N ratio and lignin content) from different cropping systems on soil C decomposition. For example, residues of legumes usually have higher content of N than wheat residues, which will influence N availability and the subsequent C decomposition. Residues with higher lignin content are also less labile and have longer residence time, potentially resulting in slower C

loss rate. **Second**, for the sensitivity assessment to climate change, we did not consider the potential fertilization effect of elevated atmospheric CO₂ concentration on crop production (Lobell and Field, 2008). This will underestimate the amount of C input into soil under climate change. The stimulated plant growth under elevated CO₂ may partially compensate the C loss under warming and/or rainfall change. **However, ignoring the effect of elevated CO₂ in our study would have limited effect on simulation results because temperature and rainfall are the two predominant factors controlling soil C decomposition, and the effect of C inputs in the scenarios caused by fertilizer reduction and residue removal were significantly larger than by impact of CO₂ change.**

The results in this study have important implications for development of effective management strategies and identification of critical zones for lasting C sequestration in agricultural soils. Sequestered soil C will be less vulnerable to management changes in drier and cooler regions, while adopting management practices that benefit soil C protection will also reduce the vulnerability. For example, although the ability of no-tillage for C sequestration is limited (Luo et al., 2010a; Powlson et al., 2014), adopting no-tillage has the advantage of reducing soil disturbance, increasing soil strength, cooling soil in summer and decreasing fresh C supply to deeper layers, and thus increase soil C stability and the resistance of sequestered C to other management changes. **Compared with response of sequestered soil C to management change**, sequestered soil C **may** respond differently to global warming. More attention should be given to cooler and wetter regions under future global warming. For rainfall change, rainfall decrease **may** generally enhance C sequestration while rainfall increase **may** generally stimulate C loss, particularly in cooler and drier regions. Increasing the effectiveness of C sequestration in the agriculture sector for long-term climate change mitigation requires reliable future climate forecast and site-specific management strategies with the consideration of local soil and climatic conditions.

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6. References

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Fig. 1. The Australian cereal-growing regions (green pixels) and the location of the 613 simulated sites (blue triangles). ACT, Australian Capital Territory; Qld, Queensland; NSW, New South Wales; NT, Northern Territory; SA, South Australia; Tas, Tasmania; Vic, Victoria; WA, Western Australia.

Fig. 2. Distribution of soil organic carbon (C) stock (0–30 cm soil layer) at the equilibrium state under optimal management (a) and the corresponding change compared with initial soil C stock (b) across the studied 613 sites before management, temperature and rainfall changes. The dashed line shows the average.

Fig. 3. Vulnerability of soil carbon (C) calculated as C loss rate to reduction of nitrogen fertilizer application (a), residue removal (b), temperature increase (c) and rainfall change (d). Boxplots show the median and interquartile range, with whiskers extending to the most extreme data point which is $1.5 \times (75-25\%)$ data range from the box.

Fig. 4. Spatial pattern for the 613 locations of soil C loss rate under 100% N application reduction (a), 100% residue removal (b), 2 °C temperature increase (c) and 20% rainfall decrease (d). Note the different scales between the panels.

Fig. 5. Coefficients for the regression model to predict soil carbon loss rate (C_{loss} , $\text{t C ha}^{-1} \text{ year}^{-1}$) under different management and climate change scenarios. The fitted model is: $\text{carbon loss rate} = \text{active organic carbon} \times (\alpha + \beta \times \text{rainfall} + \gamma \times \text{temperature})$. Values in **a**, **b** and **c** show the coefficients α , β and γ respectively, and **d** shows the determination coefficient (R^2) of the whole model. Bars show the standard error of the estimated coefficients.

Fig. 6. An example of the relationship between C loss rate and active C as impacted by temperature (a) and rainfall (b) under the scenario of 100% residue removal. The coefficients of the relationship are presented in Fig. 5.