Soil carbon change and its responses to agricultural practices in Australian agro-ecosystems: A review and synthesis

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Review

Soil carbon change and its responses to agricultural practices in Australian agro-ecosystems: A review and synthesis

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ABSTRACT

Soil is the largest reservoir of carbon (C) in the terrestrial biosphere and a slight variation in this pool could lead to substantial changes in the atmospheric CO_2 concentration, thus impact significantly on the global climate. Cultivation of natural ecosystems has led to marked decline in soil C storage, such that conservation agricultural practices (CAPs) are widely recommended as options to increase soil C storage, thereby mitigating climate change. In this review, we summarise soil C change as a result of cultivation worldwide and in Australia. We then combine the available data to examine the effects of adopting CAPs on soil C dynamics in Australian agro-ecosystems. Finally, we discuss the future research priorities related to soil C dynamics. The available data show that in Australian agro-ecosystems, cultivation has led to C loss for more than 40 years, with a total C loss of approximately 51% in the surface 0.1 m of soil. Adoption of CAPs generally increased soil C. Introducing perennial plants into rotation had the greatest potential to increase soil C by 18% compared with other CAPs. However, the same CAPs could result in different outcomes on soil C under different climate and soil combinations. No consistent trend of increase in soil C was found with the duration of CAP applications, implying that questions remain regarding long-term impact of CAPs. Most of the available data in Australia are limited to the surface 0.1 to 0.3 m of soil. Efforts are needed to investigate soil C change in deeper soil layers in order to understand the impact of crop root growth and various agricultural practices on C distribution in soil profile. Elevated atmospheric CO_2 concentration, global warming and rainfall change could all alter the C balance of agricultural soils. Because of the complexity of soil C response to management and environmental factors, a system modelling approach supported by sound experimental data would provide the most effective means to analyse the impact of different management practices and future climate change on soil C dynamics.

Contents

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1. Introduction

Soil is the largest reservoir of carbon (C) in the terrestrial biosphere. The total stores of soil organic carbon (SOC) and inorganic carbon (SIC) are more than 2100 Pg in the surface 1 m of soil, including about 1500 Pg of SOC (Post et al., 1982; Batjes, 1996). As the soil C pool is three times the amount of atmospheric C, a small variation in soil C stores could lead to marked change in the CO_2 concentration of atmosphere (Trumbore et al., 1996; Cleveland and Townsend, 2006; Davidson and Janssens, 2006). There is sufficient evidence that rapid and significant decline of SOC has occurred as a result of land use change (Post and Kwon, 2000; Guo and Gifford, 2002; Wilson et al., 2008), most notably when natural ecosystems are converted to agricultural systems. Globally, land use change and soil cultivation are estimated to have caused the loss of 136 Pg of soil C to the atmosphere since 1750 (Lal, 2004c). Davidson and Ackerman (1993) indicated that the average loss of soil C was about 40% in the plough layer to a depth of 0.3 m. However adoption of management practices like stubble retention and reduced tillage are found to potentially increase C in agricultural soils (Schlesinger, 1999; West and Post, 2002; Jarecki and Lal, 2003; Hooker et al., 2005; Valzano et al., 2005). This soil C sink capacity affects both world food security and global climate change (Lal, 2004a; Lal et al., 2007).

Decline of soil C in agro-ecosystems is mainly due to cultivation. Cultivation changes the quality and quantity of C inputs to soil and soil physical properties that affect C decomposition. It takes several decades to reach a new equilibrium of soil C after cultivation (Dalal and Mayer, 1986; West and Post, 2002; Hermle et al., 2008). Management practices aimed at avoiding soil C decline need to balance the C loss from soil with C input. The rate of soil C loss is linked to soil environment that is strongly controlled by climatic conditions. Carbon inputs to soil are mainly controlled by biomass productivity of the cropping systems, which is a function of the variable climate, soil conditions, fertilizer inputs and agronomic management. So far, there is no comprehensive assessment on the spatiotemporal dynamics of the soil C pool in Australian agro-ecosystems as affected by the interplay of those environmental and management factors (Knowles and Singh, 2003; Valzano et al., 2005).

Over the past few decades, there have been substantial changes in agricultural practice in Australia, the most notable being the adoption of Conservation Agricultural Practices (CAPs; Ugalde et al., 2007). The most commonly held view is that CAPs lead to an increase in soil C (West and Post, 2002; Jarecki and Lal, 2003; Freibauer et al., 2004). However, research results are inconclusive. Numerous studies show that long-term cultivation of soil in rainfed cropping regions decreased soil C and adoption of conservational tillage (e.g., no-till and direct drilling) only reduced the rate of soil C decline, and did not lead to soil C increase (Thompson, 1992; Dalal et al., 1995; Valzano et al., 2001b; Dalal et al., 2007). Chan et al. (2003) reported that significantly higher soil C under conservation tillage only occurred in the wetter regions (>500 mm rainfall) where soil water was not limiting plant growth. As CAPs often combine different management options such as increased crop diversity and stubble retention, information is required to understand the contribution of each available option to soil C change across different regions.

This paper reviews and synthesises available information on soil C change following cultivation and the application of CAPs in Australian agro-ecosystems. Firstly, we briefly describe the worldwide conditions of soil C status in agricultural soils and compare them with the Australian conditions. Secondly, we synthesise the research results on the effects of the application of CAPs on soil C content in Australian agro-ecosystems over time and space. This includes the impacts of the enhancement of rotation complexity, stubble retention and conservation tillage, and fertilization and irrigation. We then discuss the possible impact of future climate change (increased atmospheric CO_2 concentration, temperature and rainfall change) on soil C content. Finally, we discuss the role of system modelling in understanding soil C dynamics in agro-ecosystems and future research priorities.

2. Soil C content as affected by climate, vegetation and cultivation

2.1. On worldwide conditions

Climate and vegetation type significantly affect soil C content. Soil formation and plant growth is principally regulated by climate. Generally, soil C content is higher in wet and cool climate; it increases with increasing precipitation and decreasing temperature. For example, 58% of the 787 Pg of soil C contained in the global forest ecosystems is stored in the high-latitude forests (Dixon et al., 1994), where the climate is relatively cool and wet. Vegetation types determine the vertical distribution of soil C. On average, 33%, 42%, and 50% of the soil C up to the depth of 1 m are in the surface 0.2 m of soil under shrublands, grasslands, and forests, respectively (Jobbagy and Jackson, 2000; Ehleringer et al., 2000).

For a given bio-geographic and climatic region, C content of the soils in natural ecosystems generally reaches quasi-equilibrium between the C input and decomposition after hundreds or thousands of years, although some studies show a minor increase over time in some old virgin soils (Schlesinger, 1990; Wardle et al., 1997). However, this equilibrium can be disturbed by human activities, resulting in a marked loss of soil C to the atmosphere in a relatively short time period (Romanya et al., 2000; Post and Kwon, 2000; Guo and Gifford, 2002; Strassmann and Fischer, 2008).

Agricultural production has changed the natural ecosystems and disturbed the soil environment, leading to a significant decline of soil C, with most of C loss occurred in the first several years. Mann (1986) estimated a 20% loss of SOC following the cultivation of forests or grasslands, equivalent to approximately 1500 g m^{-2} in the surface 0.3 m of soil, with the greatest rates of change occurred in the first 20 years. It was shown that 20 years of cultivation could result in 40% reduction in soil C from the A horizon to about 0.3 m depth, with more than 50% of the loss occurring within the first 5 years (Davidson and Ackerman, 1993). The loss of different soil C fractions was largely related to management practices. For example, Chan et al. (2002) indicated tillage removed mainly particulate organic C (>53 μm) which accounted for 80% of the total C loss; whereas, stubble burning mainly resulted in the loss of mineral associated organic C (<53 μm).

Adoption of CAPs can potentially reverse or slow down the loss of SOC (Lal and Kimble, 1997; Schlesinger, 1999; Jarecki and Lal, 2003). However, the impact of CAPs largely depends on local soil types, climate conditions (e.g., temperature and rainfall), farming system, and management types (Table 1). Several studies indicate that continuous CAPs lead to increases in SOC during the first 20 to 50 years or until SOC attains a “new” equilibrium (Sauerbeck, 2001; West and Post, 2002). Lal (2004a) estimated that the world cropland soils could potentially sequester 0.4–0.8 Pg C per year by adopting CAPs. Correspondingly, this C accumulation potential in agricultural soils represents 33.3–100% of the total potential of C sequestration in world soils.

Table 1
Estimated soil C sequestration potential in agricultural soils after adoption of conservation agricultural practices in five different regions. NA, not available.

Region	Cropland area[a] (10^6 ha)	Yearly soil C sequestration potential[b] (10^{12} g C yr^{-1})	Duration[b] (year)	Average soil C sequestration potential (10^6 g C yr^{-1} ha^{-1})
Australia	48.23	17	20	0.35
China	135.36	76	84	0.56
European Union	84.78[c]	90–120	NA	1.06–1.42
India	169.70	39–49	NA	0.23–0.29
United States	179.00	83	NA	0.46

[a] Source: Food and Agricultural Organization of the United Nations (FAO). 2002. FAOSTAT on-line statistical service. Rome: FAO.
[b] Data adapted from Dalal and Chan (2001), Yan et al. (2007), Smith (2004), Lal (2004b) and Sperow et al. (2003) for Australia, China, European Union, India and United States, respectively.
[c] Including: Austria(1.479), Belgium (Not available), Denmark (2.302), Finland (2.177), France (19.515), Germany (12.038), Greece (3.87), Ireland (1.079), Italy (11.422), Luxembourg (Not available), the Netherlands (0.949), Portugal (2.705), Spain (18.530), Sweden (2.747) and the United Kingdom (5.968).

Sperow et al. (2003) suggested that U.S. cropland soils have the potential to sequester an additional 0.06–0.07 Pg C per year, on top of the present rate at 0.017 Pg C per year, through widespread adoption of soil C sequestration management practices. These practices include no-tillage, elimination of summer fallow, and maintaining winter cover. In the European Union, the overall potential soil C sequestration was estimated to be 0.09 Pg C per year (Smith, 2004; Lal, 2004c), of which about 0.023 Pg C per year was attributed to conversion to no-till management (Smith et al., 1998). No-till agriculture showed the largest potential to increase soil C sequestration compared with other CAPs such as fertilization, irrigation, improved rotation and animal manure (Smith, 2004). A one-year modelling study showed that, on average, China's croplands lost 1.6% of its SOC in the surface 0.3 m of soil just in 1990 as compared with the 0.1% of C loss in U.S. croplands (Li et al., 2003). Yan et al. (2007) estimated that the total C sequestration potential is 0.076 Pg per year in all croplands in China if all crop residues were returned to the soil and no-tillage was practiced. In India, the potential of soil C sequestration was estimated at 0.006–0.007 Pg per year under adoption of CAPs on agricultural soils (Lal, 2004b).

2.2. Australian conditions

In Australia, soil organic C is naturally low except in some eastern regions. In natural ecosystems, soil C content in the surface 0.3 m of the soil profile ranges from <10 Mg ha^{-1} in arid regions to >250 Mg ha^{-1} in relatively wet regions (i.e., coastal swamps and Tasmania), depending on specific climate and soil conditions (Webb, 2002). In water-limited regions, water availability regulates biomass production and thus the input of C to soil. In contrast, in regions where water is not limiting, radiation and temperature regulate biomass production, while temperature and other soil factors control the C decomposition rates and soil C content (Wynn et al., 2006).

Cultivation has led to a reduction in soil C in Australia. The change of soil C relative to adjacent natural ecosystems with years of cultivation is shown in Fig. 1. This figure was generated by combining data from 20 published studies across Australian agro-ecosystems (Dalal and Mayer, 1986; Bell et al., 1995; Chan et al., 1995; Cogle et al., 1995; Conteh et al., 1997; Whitbread et al., 1998; Sparrow et al., 1999; Skjemstad et al., 2001; Bruand and Gilkes, 2002; Knowles and Singh, 2003; Murphy et al., 2003; Dalal et al., 2005; Young et al., 2005; Wilson et al., 2008; Table 2). The data shows an exponential loss of soil C after the cultivation, with most loss occurring in the first 10 years (Fig. 1). The loss of soil C in the surface 0.1 m is 51% and a quasi equilibrium was reached after about 50 years of cultivation (Fig. 1A). The loss of C in the surface 0.3 m of soil was variable and ranged from 0.9% to 73.4% (Fig. 1B). This result suggests that C loss in agricultural soils mainly occurs in the surface 0.1 m of soil and is consistent with findings of other studies (Davidson and Ackerman, 1993; Murty et al., 2002; Guo and Gifford, 2002). The C loss after cultivation primarily results from reduced input of organic materials to the soil, erosion of soils offsite, downward movement of organic matter to deeper soil layers, the fracture of soil macro-aggregates, and the increased microbial activity leading to decomposition of the organic C pools.

In a review on the dynamics of soil organic matter (SOM) in rainfed cropping systems of the Australian cereal belt, Dalal and Chan (2001) showed that more than 60% of the SOM were lost from the surface 0.1 m of soil after 50 years of cultivation and cropping, a result consistent with Fig. 1A. Further, they estimated that the emission of CO_2 in Australian rainfed croplands would be reduced by more than 1.04 Pg after 20 years of adopting CAPs. The reduction is 2 times more than the current total annual emission of CO_2 for the whole of Australia (Dalal and Chan, 2001).

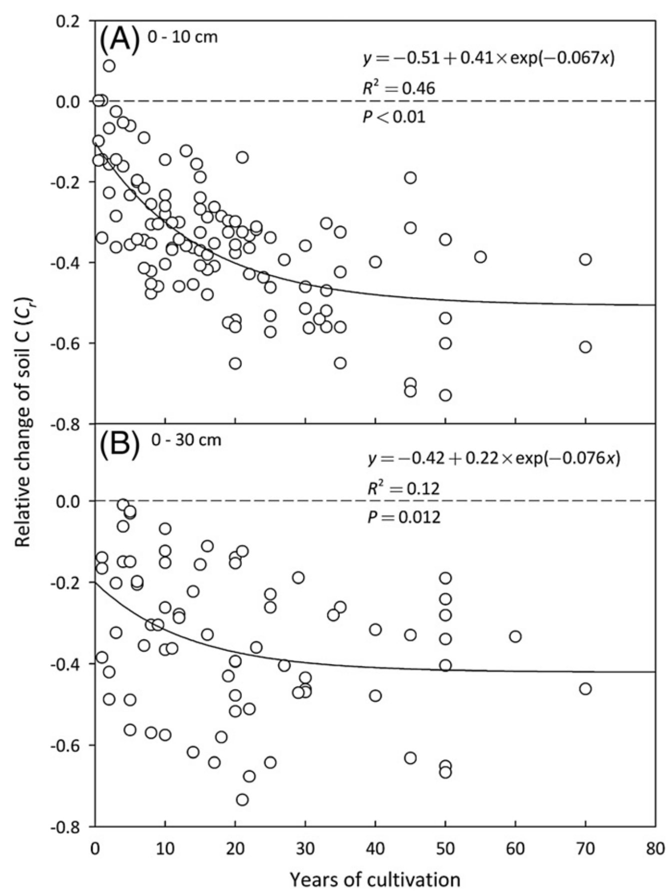

Fig. 1. The change of soil carbon relative to adjacent natural systems (C_r) in the surface 0.1 m (A) and 0.3 m (B) of Australian soils after years of cultivation.

The impacts of different agricultural practices on soil C has been extensively studied in recent years (Wells et al., 2000; Dalal and Chan, 2001; Chan et al., 2003; Knowles and Singh, 2003). Heenan et al. (1995) showed that different agricultural practices caused significant differences in the soil organic C trend over 14 years, ranging from no change to an annual loss of 400 kg ha^{-1}. The largest C loss occurred when conventional cultivation was combined with stubble burning in an annual wheat-cropping system. In a vegetable farming system, after three and a half years of adoption of conservation management including no-till and high inputs of compost with high content of organic C, SOC in conservation management systems was 75.4% higher than that in conventional management systems (Wells et al., 2000). In spite of the positive responses of soil C content to application of CAPs, some studies have indicated that soil C levels are unlikely to return to pre-cultivation levels, irrespective of the conservation agricultural practices being implemented (Whitbread et al., 1998; Bell et al., 1999).

Several studies divide Australian agricultural areas into wetter (rainfall > 500 mm) and drier regions (Chan et al., 2003; Valzano et al., 2005), and found that adoption of conservation tillage only increased

Table 2

Summary of the studies used in our analysis of the effects of conservation agricultural practices (CAPs) on soil C dynamics in Australia agro-ecosystems. R, rotation (ID, increased crop diversity; IF, increased crop frequency; IP, increased perenniality); SR, stubble retention; ZT, conservation tillage; N, N fertilizer application. NA, not available. *, carbon content at adjacent natural vegetation also was reported, which was used for data synthesis in Fig. 1.

Reference	Location	Location code	Soil type[a]	CAPs	Duration (year)	Sampling depth (m)
Armstrong et al. (1999)*	Emerald, QLD	1	Vertosol	R(ID, IP)	3	0.1
Armstrong et al. (2003)	Emerald, QLD	1	Vertosol	R(ID, IP)	7	0.1
Blair and Crocker (2000)*	Tamworth, NSW	2	Vertosol	R(ID, IP)	29	0.1
Blair et al. (1998)*	Ayr, QLD	3	Rudosol	SR	7	0.25
	Tully QLD	4	Hydrosol	SR	4	0.25
Blair (2000)	Mackay, QLD	5	Chromosol	SR	20	0.1
Blair et al. (2006a)*	Tamworth, NSW	2	Vertosol	R(ID, IP)	33	0.1
Bünemann et al. (2008)	Wagga Wagga, NSW	6	Kandosol	R(ID), SR, ZT	26	0.05
Carter and Mele (1992)	Wodonga, VIC	7	Sodosol	SR, ZT	10	0.025
Cavanagh et al. (1991)	Forbes, NSW	8	Chromosol	ZT	2	0.1
Chan and Hulugalle (1999)*	Trangie, NSW	9	Vertosol	R(ID, IF)	3	0.3
Chan et al. (1992)	Wagga Wagga, NSW	6	Kandosol	R(ID), SR, ZT	10	0.2
Chan et al. (2002)	Wagga Wagga, NSW	6	Kandosol	SR, ZT	19	0.2
Conteh et al. (1998)	Narrabri, NSW	10	Vertosol	SR, N	3	0.3
Cookson et al. (2008)	Wongan Hills, WA	11	Tenosol	ZT	6	0.1
Cotching et al. (2001)	Midlands of TA	12	Sodosol	R(IF)	7	0.15
Dalal (1989)	Warwick, QLD	13	Vertosol	SR, ZT, N	12	1.2
Dalal et al. (1991)	Warwick, QLD	13	Vertosol	SR, ZT, N	20	0.1
Dalal et al. (1995)	Warra, QLD	14	Vertosol	R(IP, ID), ZT, N	9	0.1
Dalal et al. (2007)	Warra, QLD	14	Vertosol	ZT, N	10	0.1
Fettell and Gill (1995)	Condobolin, NSW	15	Chromosol	SR, ZT, N	15	0.1
Gupta et al. (1994)	Harden, NSW	16	Kandosol	SR, ZT	6	0.15
Haines and Uren (1990)	Wodonga, VIC	7	Sodosol	SR, ZT	55	0.25
Hamblin (1984)	Merredin, WA	17	Chromosol	ZT	6	0.25
Heenan et al. (1995)	Wagga Wagga, NSW	6	Kandosol	R(ID), SR, ZT	15	0.1
Holford et al. (1998)	Tamworth, NSW	2	Kandosol	R(IP ID)	27	0.15
Hoyle and Murphy (2006)	Merredin, WA	17	Chromosol	SR	16	0.05
Hoyle et al. (2006)	Merredin, WA	17	Chromosol	SR	17	0.05
Hulugalle (2000)	Narrabri, NSW	10	Vertosol	R(IF), ZT	13	0.6
Hulugalle and Entwistle (1997)	Narrabri, NSW	10	Vertosol	R(ID, IF), ZT	9	0.6
Hulugalle et al. (1997)	Narrabri, NSW	10	Vertosol	R(ID, IF), ZT	10	0.6
Hulugalle et al. (2006)	Warren, NSW	18	Vertosol	R(ID, IF)	3	0.6
Hulugalle et al. (2007)	Warra, QLD	14	Vertosol	R(ID, IF)	11	0.6
Loch and Coughlan (1984)	Warwick, QLD	13	Vertosol	SR, ZT	5	0.1
Mason (1992)	Merredin, WA	17	Kandosol	R(ID), SR	9	0.1
	Nabawa, WA	19	Chromosol	SR	10	0.1
	Wongan Hills, WA	11	Kandosol	SR	10	0.1
Noble et al. (2003)	Tully QLD	4	Dermosol	R(IP), SR	6	0.1
Packer and Hamilton (1993)	Cowra, NSW	20	Chromosol	ZT	7	0.1
	Grenfell, NSW	21	Chromosol	ZT	6	0.1
Pankhurst et al. (2002a)	Cowra, NSW	20	Chromosol	SR, ZT	17	0.1
Pankhurst et al. (2002b)	Cowra, NSW	20	Chromosol	SR, ZT	5	0.1
	Harden, NSW	16	Chromosol	SR, ZT	5	0.1
Rahman et al. (2007)	Wagga Wagga, NSW	6	Kandosol	R(IP), SR, ZT	22	0.1
Robertson and Thorburn (2007)	Harwood, NSW	22	NA	SR	1	0.05
	Mackay, QLD	5	Chromosol	SR, ZT	1	0.25
	Tully QLD	4	Hydrosol	SR	1	0.05
Smettem et al. (1992)	Kapunda, SA	23	Sodosol	R(ID), ZT	8	0.05
Standley et al. (1990)	Biloela, QLD	24	Vertosol	SR, ZT	7	0.1
Thompson (1992)	Warwick, QLD	13	Vertosol	SR, ZT, N	11	0.125
Valzano et al. (2001a)	Natimuk, VIC	25	Vertosol	SR	2.5	0.225
Valzano et al. (2001b)	Peak Hill, NSW	26	Sodosol	ZT	3	0.1
Wang et al. (2004)	Warwick, QLD	13	Vertosol	SR, ZT, N	33	0.1
Whitbread et al. (2003)*	Warialda, NSW	27	Dermosol	R(ID, IP)	7	0.05
White (1990)	Avondale, WA	28	Chromosol	ZT	9	0.25
	Merredin, WA	17	Chromosol	ZT	9	0.25
	Wongan Hills, WA	11	Tenosol	ZT	9	0.25
Willis et al. (1997)	Trangie, NSW	9	Sodosol	R(IF), ZT	4	0.15

[a] Soil classification based on Isbell (2002).

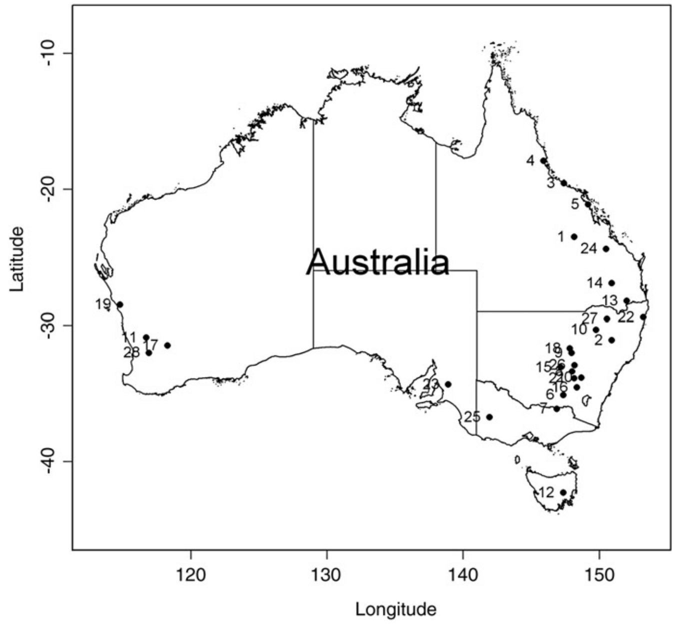

Fig. 2. Location of the studies in Australia. Numbers show the location code as listed in Table 2.

SOC in wetter regions, but could not reverse the decline of SOC of croplands in drier regions. Further, the combination of various agricultural practices makes it difficult to analyse the effects of each individual option.

3. Soil C content change in Australian agro-ecosystems after adopting conservation agricultural practices

We mainly focus on three major types of CAPs: i) cropping systems and rotation, ii) stubble management and tillage, and iii) the application of irrigation and fertilizer. The major studies are given in Table 2 and their locations shown in Fig. 2. For each type and study, we calculated the relative change of soil C under CAPs relative to conventional agricultural practices in paired experiments (i.e., other conditions and durations were kept similar and the same duration of practices). The relative change of soil C (C_r) was calculated as:

$$C_r = \frac{C_{CAPs} - C_{Conventional}}{C_{Conventional}}, \qquad (1)$$

where C_{CAPs} is the soil C content under the conservation agricultural practices (CAPs), $C_{Conventional}$ is the soil C content under the conventional agricultural practices. Positive C_r indicates an increase of soil C after adoption of CAPs; negative C_r indicates a decline of soil C after adoption of CAPs.

Most of the studies only presented the C content as fraction (i.e., g C per kg soil, g C per 100 g soil or mg C per g soil), we standardized the soil C content as percentage (C_c) (%, i.e., g C per 100 g soil). Several studies reported the soil C mass (C_m, e.g., t C ha^{-1} and kg C m^{-2}), we recalculated the soil C content as:

$$C_c = \frac{C_m}{BD \times D}, \qquad (2)$$

where BD is the soil bulk density, D is the soil depth.

3.1. Crop systems and rotation

Many studies suggest that increasing rotation complexity could potentially lead to soil C increases in agricultural soils (Follett, 2001; West and Post, 2002; Jarecki and Lal, 2003; Bremer et al., 2008). This includes changes from monoculture to continuous rotation cropping, from crop-fallow systems to continuous systems and an increase in crop diversity in a rotation (West and Post, 2002). Synthesizing a global database from 67 long-term agricultural experiments, West and Post (2002) concluded that increasing crop diversity and/or excluding long-fallow periods could result in significant accumulation of SOC attaining a "new" equilibrium after about 40–60 years.

Pooling all data together, the relative changes of soil C content (C_r) calculated from 23 published studies in Australia (Table 2) are shown in Fig. 3. The data include 211 observations over a 33 year period, concerning the effects of enhanced crop diversity, cropping frequency and perenniality on soil C content in Australian agro-ecosystems. Soil C content increased in approximately 86% of all reported observations; the relative change of soil C content ranged from −11.8% to 118.3%, with an average value of 9.9% (Fig. 3). We sorted the data into 5-year intervals from 0 to 35 years based on the duration of the practice. After excluding the outliers using *Q*-test, paired *t*-test indicated that soil C content significantly increased ($P<0.01$) for all duration intervals. However, the level of relative change in soil C content did not show a significant difference ($F_{(5,\ 205)}=1.71$, $P=0.13$) among duration intervals (Fig. 3). Our result is different from the global data analysis performed using a quantile regression by West and Post (2002). Their study showed a gradual increase in soil C content until reaching a quasi equilibrium after 40–60 years with increasing rotation complexity. To quantify the effects of conservation cropping practices on soil C accumulation, West and Post (2002) used the mean difference between the soil C content in the first year of application of conventional practices and the soil C content in the latest year since adopting conservation practices. Their analysis method assumed that the soil C content under the application of conventional practices was constant and dismisses the annual dynamic features of soil C caused by environment and management practices. Our method of analysis was based on paired experiments that had the same duration of the application of conventional and conservation

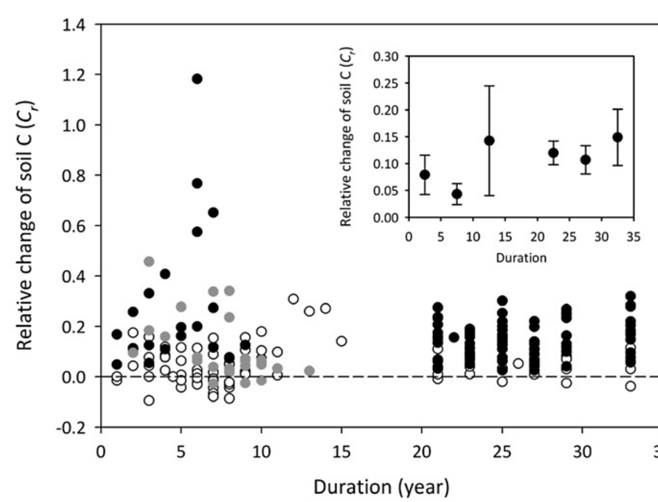

Fig. 3. The relative changes of soil C (C_r) in agricultural soils after enhancement of crop complexity (increased crop diversity, cropping frequency and perenniality). Soil depth range from 0.05 to 0.6 m, and are not normalized to a specific depth. Open, gray and filled circles show the values under the cropping systems of one crop per year, two or more crops per year and rotation with perennial plants, respectively. Inset, the average change of soil C content after sorting data into 5-year intervals, and bars show the 95% confidence interval. The largest two values are outliers (*Q*-test) and are excluded in the calculation of the average and the 95% confidence interval of the relative change of soil C content. See details in text.

practices, and overcomes this problem. The results suggest that soil C content increased in the first several years after increasing crop complexity and the value then remained relatively stable, a finding which is consistent with that of Bremer et al. (2008).

The variations in the Australian results on the impact of increasing rotation complexity arise from the mix of different practices involved (Blair and Crocker, 2000; Noble et al., 2003; Armstrong et al., 2003; Hulugalle et al., 2007; Bünemann et al., 2008; Hulugalle and Scott, 2008). Here we group rotation complexity into three categories: 1) increased crop diversity (ID) referring to a change from continuous monoculture to continuous rotation, 2) increased cropping frequency (IF), i.e., a change from one crop per year to two or more crops per year (e.g., from continuous wheat to wheat–cotton double cropping), and 3) increased perenniality (IP), i.e., a change from annual crops to a rotation with perennial crops. To assess the effects of nitrogen-fixing plants on soil C content, we also separate rotation systems with and without legume crops (e.g., wheat–cotton rotation vs. wheat–chickpea rotation). Further, we analysed the data between systems with and without a fallow period.

Comparing with monoculture (with or without long-fallow) as the baseline, increasing crop frequency and perenniality led to significant increase in soil C ($F_{(2,209)}=22.72$, $P<0.01$; Fig. 4A). Enhancing crop diversity only resulted in 5.3% increase in soil C. Changing from one to two crops per year nearly doubled the soil C increase (10.1%). Introducing perennial plants into rotation led to significantly higher soil C increase of 17.8% ($P<0.05$); whereas, introducing annual legumes into rotation was comparable with introducing non-legumes into rotation (Fig. 4B). Introducing perennial plants into rotation resulted in significant accumulation of soil C, especially for contiuous cropping with a fallow. Overall, rotation with perennial plants led to significantly more C accumulation in soil than other cropping practices regardless of the fallow managements (Fig. 4C and D).

These results suggest that increase in land cover by either increasing cropping frequency or growing perennial plants will have the most impact to increase soil C accumulation. This can be explained by the greater total production of both above- and below-ground biomass. Moreover, growing perennial plants potentially reduces the disturbance of soil and soil C loss through erosion or leaching. Some studies suggest that introduction of nitrogen-fixing plants enhances organic matter input and N pool by symbiotically fixed N, which is beneficial to soil quality and the succeeding crop, thus increasing the C input into the soil (Willis et al., 1997; Whitbread et al., 1998; Armstrong et al., 1999; Blair et al., 2006b). However, our analysis indicates that introducing annual legumes into rotation does not stimulate more C than other cropping practices unless introducing perennial legumes, such as lucerne (Fig. 4B).

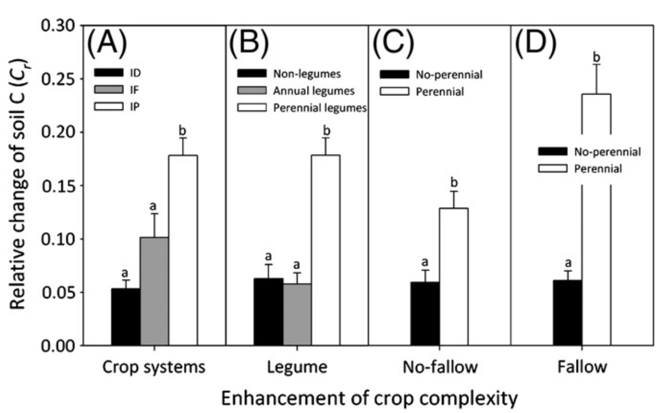

Fig. 4. The effects of enhancement of crop complexity on the relative changes of soil C (C_r). A: ID, increased crop diversity; IF, increased crop frequency; IP, increased perenniality. B: Non-legume, rotation not including legumes; Annual legumes, rotation with annual legumes; Perennial legumes, rotation with perennial legumes. C: Based on conventional croplands without a fallow period. D: Based on conventional croplands with a fallow period. No-perennial, no perennial plants introduced into rotation; perennial, introducing perennial legumes into rotation. Bars show the standard error. Different letters indicate the significant effects at the level $P<0.05$. More details in the text.

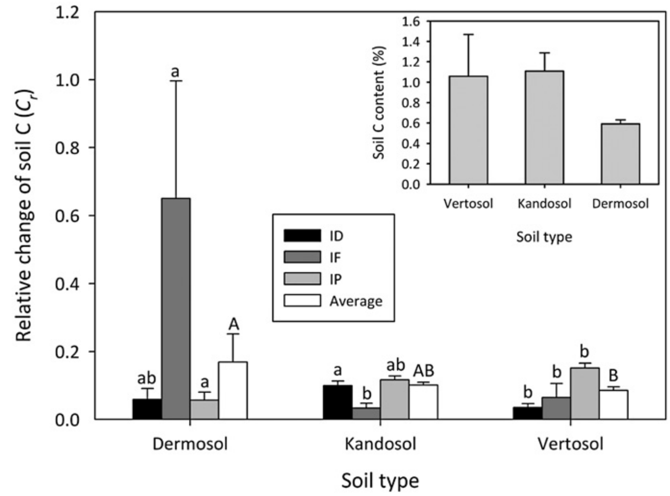

Fig. 5. The effects of soil types on the relative changes of soil C (C_r) under three different cropping types. Data are not normalized to a specific soil depth. Inset, the average soil C content (%, g C per 100 g soil) in the three corresponding soil types under the application of conventional cropping practices ($C_{Conventional}$). Average, the average values under the three cropping systems (i.e., ID, increased crop diversity; IF, increased crop frequency; IP, increased perenniality). Vertical bars represent the standard error. For the same cropping types, different letters indicate the significant effects at the level $P<0.05$. See details in text.

Fig. 5 shows the soil C change after adopting different CAPs on different soils. Based on the current data, increased crop diversity (ID) led to an increase in soil C content by 10% in Kandosol, 6% in Dermosol and 3.5% in Vertosol. Increased crop frequency had the greatest impact on soil C content in Dermosol (increase by 65%, $n=3$), while increased perenniality had the greatest impact on Vertosol (15.2%). On average, soil C content increased by 16.9% in Dermosol soils, which is markedly higher than the 8.5% and 10.3% increases in Vertosol and Kandosol soils, respectively ($F_{(2,\ 200)}=2.99$, $P=0.051$). However, different soils have distinct baselines of soil C content under the conventional agricultural practices (i.e., $C_{Conventional}$; Fig. 5) and the limited datasets do not allow for detailed analysis to trace the exact causes for such changes. As the soil C baseline in Dermosol is significantly lower than that in the other two soils, the absolute amount of soil C accumulated in Dermosol was similar to that in other two soil types.

3.2. Tillage and stubble management

Tillage and stubble management significantly affect the soil C content of agricultural soils. Major studies in Australia on the effects of these two management options on soil C dynamics include Bünemann et al. (2008), Chan et al. (2002), Conteh et al. (1998), Dalal and Mayer (1986), Hoyle et al. (2006), Pankhurst et al. (2002b), Valzano et al. (2005) and Wang et al. (2004).

Tillage disrupts soil aggregation, mixes the different soil particles, recycles nutrients, and redistributes the biomass C in soil profile. In general it is found to result in a decline of soil C (Roberts and Chan, 1990; Six et al., 2000b; Bronick and Lal, 2005). Two hypotheses have been put forward to explain the underlying mechanism. Firstly, tillage fragments macro-aggregates and increases the surface area for soil microbes to attack and decompose the originally physically aggregate-protected soil C (Beare et al., 1997; Six et al., 1999, 2000a; Mikha and Rice, 2004). Secondly, tillage incorporates aboveground fresh organic matter into soil, which provides nutrients and energy for microbial growth and therefore stimulates the decomposition of soil C including inert organic C (Fontaine and Barot, 2005; Fontaine et al., 2007).

Stubble burning converts biomass C into CO_2 and black C which is similar to charcoal. It generally leads to a reduction of soil C. The high temperatures generated by fire affects microbial activity in the surface soil, alters soil structure and soil hydraulic properties (Valzano et al., 1997; Kumar and Goh, 2000). The remaining black C is resistant to microbial decomposition and can persist in the soil for centuries (Harden et al., 2000). For this reason, black carbon has been proposed as a method to store C and offset the anthropogenic emission of CO_2 (Marris, 2006; Lehmann, 2007). Black C, on the other hand, could stimulate microbial growth and activity (Zackrisson et al., 1996; Pietikäinen et al., 2000), which may indirectly influence soil C dynamics. For example, in a ten years trial, Wardle et al. (2008) found that fire-derived charcoal promoted loss of native C (humus) in boreal forest soils.

Conservation tillage and stubble retention may prevent soil degradation and C loss through minimising soil disturbance, increasing the input of biomass C to the soil, and reducing the decomposition and removal of biomass C from crop land (Schlesinger, 1999; Jarecki and Lal, 2003; Antle et al., 2007; Lal et al., 2007). Replacing conventional tillage with conservation tillage (e.g., no-till, and direct-drill) has been reported to improve soil conditions and significantly increase soil C content (Dalal et al., 1991; Cavanagh et al., 1991; Chan et al., 2002; Pankhurst et al., 2002b; Valzano et al., 2005; Rahman et al., 2007). Similarly, adoption of stubble retention practices reduces the export of biomass C from the soil–plant system, and generally increases soil C (Hoyle and Murphy, 2006; Robertson and Thorburn, 2007).

Fig. 6 summarises the data from 39 published articles (Table 2) on the impacts of stubble retention and/or conservation tillage on soil C change in the surface 0.1 m of soil ($F_{(4,\ 285)} = 2.67$, $P < 0.05$). On average, changing from conventional to conservation tillage increased soil C content by 9.6%. Retaining stubbles rather than stubble burning resulted in an increase in soil C by 10.2%. Combining stubble retention and conservation tillage increased soil C content by 16.37% as compared with stubble burning and conventional tillage. The impact of combining conservation tillage and stubble retention is significantly higher than the separate application of these two practices, with 5.78% for stubble retention and 2.96% for conservation tillage only (Fig. 6).

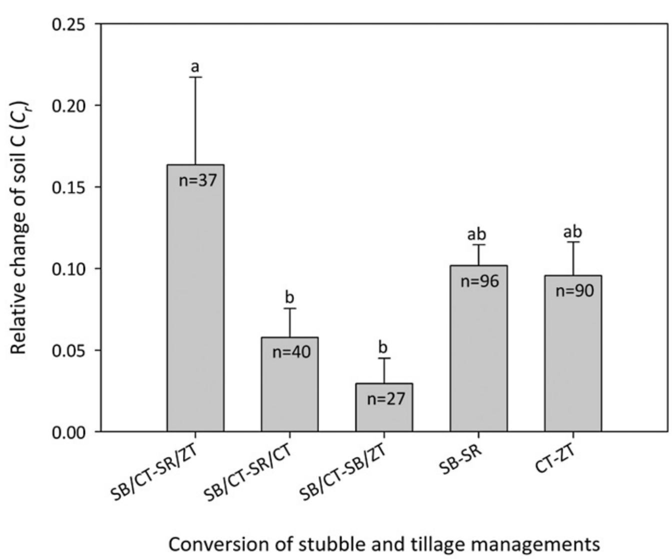

Fig. 6. The relative changes of soil C (C_r) in the surface 0.1 m of soil after adoption of different conservation agricultural managements. A: SB/CT–SR/ZT, conversion from stubble burning (SB) and conventional tillage (CT) to stubble retention (SR) and zero tillage (ZT); B: SB/CT–SR/CT, conversion from SB and CT to SR and CT; C: SB/CT–SB/ZT, conversion from SB and CT to SB and ZT; D: SB–SR, conversion from SB to SR; E: CT–ZT, conversion from CT to ZT. Vertical bars represent the standard error. Values followed by the same letter are not significantly different ($P < 0.05$).

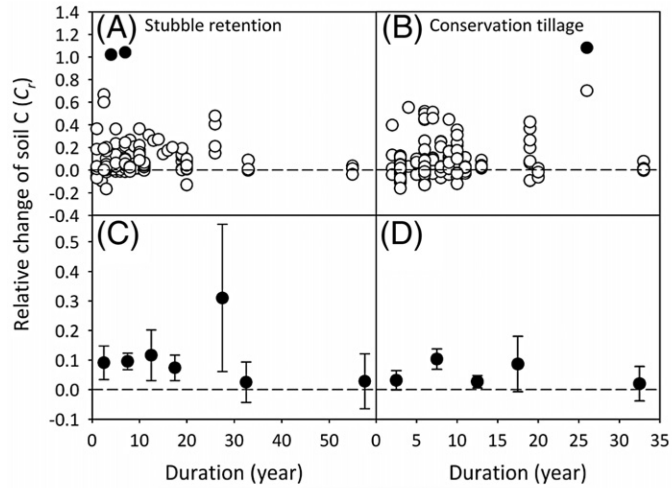

Fig. 7. The relative changes of soil C (C_r) after adoption of stubble retention (A) and conservation tillage (B). Soil depth range from 0.05 to 0.6 m, and are not normalized to a specific depth. The average change of soil C after adoption of stubble retention and conservation tillage are shown in (C) and (D), respectively, after categorizing data into duration intervals of five years. Bars show the 95% confidence interval. Solid circles in (A) and (B) show the outliers (*Q*-test) and are excluded when calculating the 95% confidence intervals in (C) and (D).

Adoption of conservation stubble and tillage managements increased soil C content in 84.0% ($n = 119$) and 71.7% ($n = 159$) of reported observations (Fig. 7A and B). The change in soil C ranged from −16.7 to 104.1% after adoption of stubble retention (Fig. 7A) and from −16.3 to 107.9% after adoption of conservation tillage (Fig. 7B) with an average of 8.0% and 11.1%, respectively. Grouping data into 5-year time interval after the adoption of the conservation methods and excluding outliers (*Q*-test), our analysis shows that adoption of stubble retention or incorporation significantly increased soil C content only in the first 25 years. Soil C was increased by 9.1% in 0–5 years, 9.5% in 5–10 years, 11.6% in 10–15 years, 7.3% in 15–20 years, and 31% in 20–25 years (Fig. 7C). The change of soil C content showed significant differences among duration intervals under the adoption of conservation stubble managements ($F_{(6,\ 110)} = 2.60$, $P < 0.05$; Fig. 7A). For conservation tillage, soil C content was significantly increased only during the periods of 5–10 and 10–15 years with increases of 10.3% and 2.6%, respectively. The change of soil C content also varied significantly among the duration intervals under the adoption of conservation tillage ($F_{(4,\ 152)} = 2.76$, $P < 0.05$; Fig. 7D). Although soil C content showed significant differences among duration intervals, there was no apparent relationship between the magnitude of soil C change and the duration of both conservation stubble and tillage managements (Fig. 7). Both stubble retention and conservation tillage did not seem to lead to significant change in soil C content in long-term. This may be attributable to the relatively small ($n < 4$) sampling size (Fig. 7) and emphasizes the need for longer term studies to investigate the long-term impact of these two practices on soil C content.

The effects of adoption of conservation stubble and tillage on potential soil C sequestration were significantly dependent on soil types ($F_{(3,\ 273)} = 26.90$, $P < 0.001$; Fig. 8). Soil C content was increased by 26% on Kandosol, which was significantly higher than the 6.31% and 11.82% measured on Sodosol and Chromosol, respectively (Fig. 8). As Kandosol also had the relatively higher baseline soil C content, it had the greatest increase in soil C among the four soil types. Vertosol showed the least C increase of 3.3% following the adoption of conservation management. However, Vertosol had the largest soil C baseline (Fig. 8), which made the absolute amount of soil C accumulation comparable with Sodosol and Chromosol. The difference in the relative change of soil C between soil types may attribute to many factors, which may be either climate related ones affecting

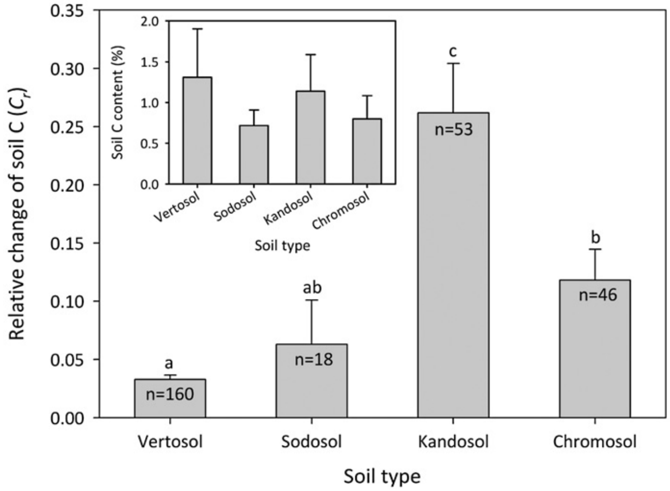

Fig. 8. The relative changes of soil C (C_r) in the surface 0.1 m of soil after adoption of conservation tillage and stubble retention in four soil types. Inset, the average soil C content (%, g C per 100 g soil) in corresponding four soil types under the application of conventional stubble and tillage practices ($C_{Conventional}$). Vertical bars represent the standard error. Values followed by the same letter are not significantly different (P<0.05).

productivity, or soil related ones affecting C decomposition (White, 1990; Mason, 1992; Robertson and Thorburn, 2007).

Rainfall or soil water balance is a critical factor that has significant effects on potential soil C accumulation under conservation tillage and stubble managements (Fig. 9). Based on the available data (Table 2), adoption of conservation tillage led to the greatest increase in soil C (38.6%) in the region with 500–600 mm annual rainfall; whereas, in region with annual rainfall less than 300–400 mm or greater than 600 mm, the increase of soil C content was significantly lower (Fig. 9). For stubble management, soil C content had the largest increase (25.3%) in the regions with 300–400 mm rainfall when the stubble management changed from burning to retention. This result may attribute to the lower production of residues in low rainfall areas and the higher content and decomposition of soil C in high rainfall areas, which minimise the overall change of soil C balance between input and output.

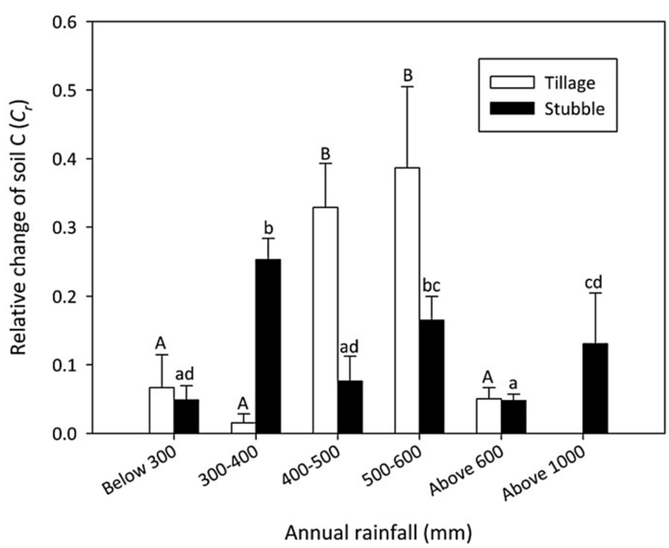

Fig. 9. The effects of rainfall on the relative changes of soil C (C_r) in the surface 0.1 m of soil under conservation tillage and stubble retention managements. Vertical bars represent the standard error. Values followed by the same letter are not significantly different (P<0.05). The capital letters show significance of tillage management, while lower case letters apply to stubble management.

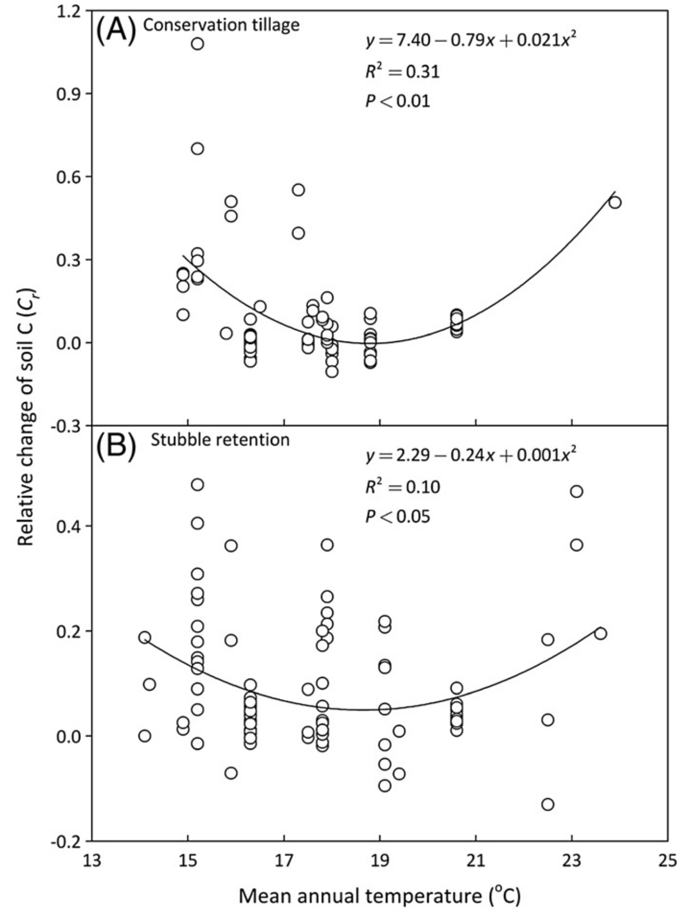

Fig. 10. The effects of temperature on the relative changes of soil C (C_r) in the surface 0.1 m of soil managed using conservation tillage (A) and stubble retention (B).

Temperature also plays a role in soil C change after adoption of conservation tillage and stubble managements via its impact on decomposition (Fig. 10). Generally, soil C content had the smallest increase in the regions with an average annual temperature of 18–19 °C. In low temperature areas, the increased C input into soil under conservation stubble and/or tillage managements decompose at a slower rate than in high temperature areas, and consequently relatively more C may remain in the soil at a given time scale. In high temperature regions, where rainfall is usually higher in Australia, there may be much higher C input into soil from high productivity. However, the interaction between rainfall and temperature needs to be analysed in a more systematic approach (see later).

3.3. Fertilization and irrigation

Irrigation and fertilization increase crop productivity in areas with water and nutrient deficiencies, thus have the potential to increase soil C content through increasing C inputs. However, the impact of either irrigation or fertilization is similar and depends on whether water or nutrient is limiting. In a water-limited condition, increasing nutrient inputs do not lead to greater production. This is also true for increasing water input through irrigation in nutrient-limited soils.

In Australia, the response of soil C change to fertilization is largely dependent on available water supply to the crops. Fig. 11 synthesises the data from 8 published studies (Table 2) on the relative change of soil C content as affected by the amount of N fertilizer application ($F_{(4, 70)} = 16.27$, P<0.001). In general, higher N input occurs mostly in wet areas, therefore, on average, the change in soil C increases with N input levels. However, there is no apparent relationship between the soil C change and the duration of N fertilizer application (Fig. 12). The large variability

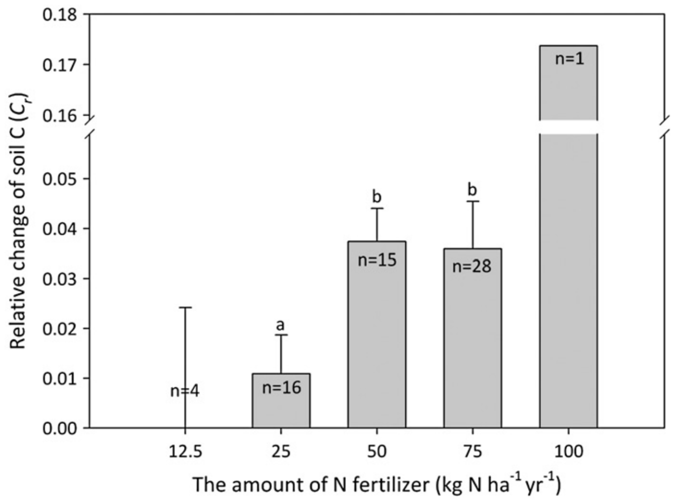

Fig. 11. The relative changes of soil C (C_r) under different levels of N fertilizer application. The treatment of 25 kg N ha^{-1} yr^{-1} includes the application of 23 and 25 kg N ha^{-1} yr^{-1}, the 50 kg N yr^{-1} ha^{-1} series includes the application of 46.7 and 50 kg N ha^{-1} yr^{-1}; and the 75 kg N ha^{-1} yr^{-1} series includes the application of 69 and 75 kg N ha^{-1} yr^{-1}. The data is not normalized to a specific soil depth. Vertical bars show the standard error ($P<0.05$) and values followed by the same letter are not significantly different ($P<0.05$).

of the soil C content change may be the result of differences in many other factors, such as water availability and stubble management for example (Dalal, 1989; Thompson, 1992; Heenan et al., 1995; Wang et al., 2004; White, 1990). The contribution of fertilization to the increase of below-ground biomass, especially roots, has been suggested to be insignificant in agricultural soils (Skjemstad et al., 1994; Dalal et al., 1995; Heenan et al., 1995). N fertilizer has been shown to promote the decomposition of crop residues and soil C (Neff et al., 2002; Khan et al., 2007), which also offset the possible increasing C input as crop residues.

The application of various types of fertilizers may complicate the influence of fertilization on soil C dynamics. Many organic fertilizers themselves contain C. The application of greenwaste, biochar and manure, could significantly increase soil C in the short-term (Chan et al., 2008a,b). However, Chan et al. (2007) pointed out that the long-term effects of biochar application need to be confirmed and quantified. Microbial community activity and structure are sensitive to soil nutrient condition. Fertilization, regardless of its type, could influence microbial

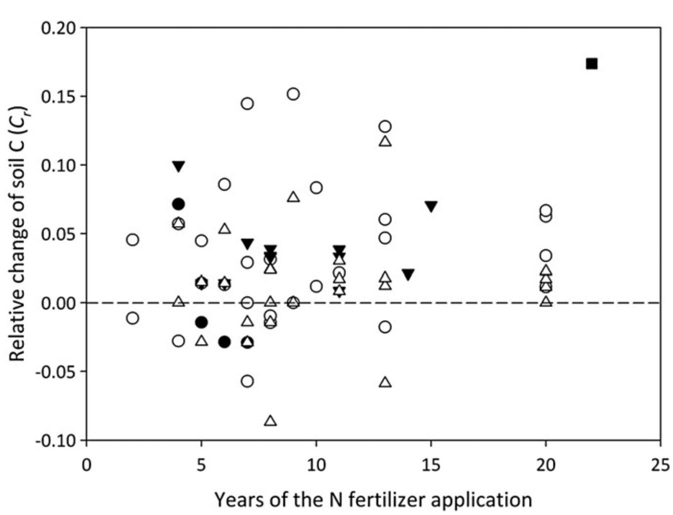

Fig. 12. Relative change of soil C (C_r) with time for treatment that received 12.5 kg N ha^{-1} yr^{-1} (●); 25 kg N ha^{-1} yr^{-1} (○); 50 kg N ha^{-1} yr^{-1} (▼); 75 kg N ha^{-1} yr^{-1} (△); 100 kg N ha^{-1} yr^{-1} (■). Data was not normalized to a specific soil depth.

community structure for several years (Cookson et al., 2005), which would also affect soil C decomposition.

There is little information available on the effects of irrigation on long-term soil C dynamics in Australian agro-ecosystems. Most cropping systems in Australia are water-limited, and irrigation improves crop yields and increase biomass C, but whether the increased C would return to soil largely depends on other agricultural management practices (Willis et al., 1997). Moreover, irrigation usually induces frequent wetting and drying, which impacts many soil processes such as cracking, residue decomposition and mineralization, and soil mixing and inversion. Chan and Hulugalle (1999) suggested that irrigation stimulates SOC decomposition, which led to faster loss of soil C than in non-irrigated soils.

4. Discussion and future research priorities

4.1. Synthesis of existing data

Through review of literature and synthesis of existing data, we found that soil C content decreased exponentially after cultivation in Australian agro-ecosystems and reached a new equilibrium with the loss of about 51% of C in surface 0.1 m of soil after 40–50 years. In general, adoption of CAPs led to carbon increase in agricultural soils. However, there remain large variations among the effects of various CAPs over time and space. Increasing crop frequency and perenniality and combination of stubble retention and conservation tillage contributed most to increased soil C accumulation. However, based on the available data, no consistent trend of increase in soil C was found with the duration of CAP applications, implying that a question remains regarding long-term impact of CAPs. The impacts of fertilization and irrigation are largely dependent on climatic regions and how the crop stubble is handled. The interplay between climate, soil, cropping and management systems determines the carbon productivity and both the input and output of C in soil, and thus determines the direction of change in soil C.

Our analysis was largely based on the relative change of total soil C or organic C after conversion of agricultural practices. We did not differ these two C terms. However, by analysing the difference between the relative change of total soil C and organic C (based on data listed in Table 2), we found that the change of the two C terms did not show significant differences after cultivation.

Recently, some authors questioned the sampling of only the top soils in assessing the impact of conservation tillage (Baker et al., 2007; Lal, 2009). Several studies suggest that different tillage options just redistribute C differently in the soil profile (Blanco-Canqui and Lal, 2008; Poirier et al., 2009). Zero tillage results in more C being distributed in the top surface soil. Although this may have a positive effect to improve soil quality in the surface layer, it does not change the overall C content in the whole soil profile. Moreover, the large variability in the published data in terms of impact of CAPs may also be a result of too shallow sampling depth in soil. More studies including C changes in deeper soil layers to a depth of 1 m or the depth of crop rooting zone are needed in order to get a full picture of the impact of different cropping and management systems.

In summary, the long-term impact of CAPs on soil C change is still inconclusive. Firstly, most of the studies are based on a limited number of experiments conducted at specific locations (climate and soil combinations) and in a relatively short period. The results may not extrapolate well to other agro-ecological regions. Secondly, inexplicit separation of different management options makes it difficult to analyse the impact of individual options. Thirdly, an experimental approach provides valuable data, but is always limited by available resources. It is impractical to expect an experimental study to cover the major possible combinations of climate, soil, cropping and management systems. Thus, it may not be able to produce a complete picture of the interplay between these systems as they impact on soil carbon change. Additionally, future climate change will alter both aboveground and below-ground C processes, and then the soil C dynamics in agro-ecosystems. As pointed out by Giller et al. (2009), there is an urgent need for critical assessment under which ecological and socio-economic conditions what CAPs are best suited. A system modelling approach based on sound understanding of processes in the soil–plant–atmosphere system may provide an effective means to explore the impact of the complex interactions on soil carbon change.

4.2. Impacts of future climate change

Globally, field studies have found that increasing atmospheric CO_2 leads to higher C assimilation by plants (Zak et al., 1993; Ainsworth and Long, 2005), and results in increased litter accumulation in natural systems (Liu et al., 2009) and crop residues in agro-ecosystems (Torbert et al., 2000), thus a higher C storage in soils (Jastrow et al., 2005). Prior et al. (2005) compared the elevated CO_2 (683 ppm_v CO_2) effects on biomass production and soil C in conventional and conservation cropping systems, and found that elevated CO_2 increased the amount of crop residue (but the response was strongly crop-type-dependent) and C concentration in the top 0.05 m soil by 44% compared with at ambient CO_2 in the conservation treatment after four years. Although several studies showed enhancement on crop root growth by elevated CO_2 (Chaudhuri et al., 1990; Prior et al., 1994), this enhancement remains inconclusive under different agricultural management practices. The net effect of elevated CO_2 will depend on the balance between water, and nutrient availability that controls plant growth and microbial activity that decomposes C (van Groenigen et al., 2006; Luo et al., 2006).

Temperature change or fluctuation can significantly affect how elements cycle through litter and soil (Anderson, 1991; Hobbie, 1996; Cornelissen et al., 2007; Dang et al., 2009). Fuhrer (2003) reviewed many aspects of global warming impacts including possible reduction of nutrient use efficiency and increase of crop water consumption. These negative impacts may reduce or reverse the beneficial effects of elevated CO_2 on biomass production and reduce C input into soil. Changes in rainfall will significantly influence soil water dynamics and other water-related ecosystem processes. In grassland, Chou et al. (2008) and Knapp et al. (2002) found that increased rainfall caused significant soil C loss, although aboveground net primary productivity (ANPP) was significantly increased. However, C cycling may be less sensitive to changes in total precipitation and more affected by rainfall fluctuation. Microbial mediated processes, such as C and N mineralization and soil CO_2 flux, can respond quickly to small rainfall events. For example, Harper et al. (2005) found that seasonal mean soil CO_2 flux decreased by 8% under reduced rainfall amounts (reduced by 30%), by 13% under altered rainfall timing (50% increase in length of dry intervals between events), and by 20% when both were combined in tallgrass prairie comparing with ambient rainfall. The change of soil CO_2 flux indirectly suggests that soil microbes and/or roots are impacted by rainfall change and highlights the importance of soil water dynamics in regulating soil C cycle.

In general, there still lacks information on the effects of possible climate change on soil C processes in agro-ecosystems not only in Australia but also all over the world. Our data synthesis indicates that the impacts of agricultural managements on soil C stock vary with climatic conditions, i.e., temperature and rainfall. Further studies combining agricultural practices with climate change are warranted.

4.3. Role of systems modelling

Soil C content and its dynamics are influenced by complex interactions between climate, soil, cropping, and management in agricultural systems. The combination of climate and soil often determines the potential productivity of plant or cropping systems. Together with a given type of management system, it also decides the level of C input (root, residue, and manure application) into and output (decomposition and other C losses) from the soil. Therefore, the potential C storage capacity of the soil will be dependent on the primary productivity limited by climate, soil and management.

Agricultural systems models capture the dynamic interplay between climate, soil and cropping systems, and provide an effective means to evaluate the impact of management intervention on the dynamics of the soil–plant system. Agricultural systems modelling facilitates scenario analyses by specifying different combinations of management options, and the investigation of the impact of long-term climate variability and future climate change on agricultural systems productivity and soil C dynamics. An agricultural systems model with well-tested crop and soil modules enables the exploration of the interactions among agricultural practices in regulating soil C dynamics (Thomas et al., 1995; Heenan et al., 2004; Hooker et al., 2005; Al-Kaisi et al., 2005; Wang and Dalal, 2006; Poirier et al., 2009) across space and time, which is otherwise impractical through conducting field experiment because of the large spatiotemporal variability in both climate and soils and the possible management options.

Modelling methods have been used to simulate the effects of agricultural managements on soil C dynamics and assess the potential capacity of C sequestration under conservation agricultural practices in some places (Kucharik et al., 2001; Wang et al., 2008; Qiu et al., 2009). In Australia, the CENTURY model was used to model continuous cultivation and cereal cropping systems (Carter et al., 1993; Chilcott et al., 2007) and was found to satisfactorily predict the impact of long-term cultivation and cereal cropping on total organic C as well as other related attributes. Models have been used to assess soil C dynamics under future climate change, such as CO_2 increase, warming, rainfall variability and their combination elsewhere. For example, Grace et al. (2006) used the SOCRATES model to predict SOC stores of the North Central Region of USA by the year 2100, with temperature and precipitation increasing by an average of 3.9 °C and 8.1 cm, respectively. They found that SOC stores would decline by 11.5 and 2% (in relation to 1990 values) for conventional and conservation tillage scenarios, respectively.

However, no comprehensive modelling studies are available to show the effect of variations in soil and climatic conditions on the capacity of the soil to store C. In spite of the experimental studies reviewed in the paper, there is a lack of modelling investigation on the impact of conservation agricultural practices, their combinations and future climate change on soil C dynamics across different agro-ecological regions. Such studies are needed to determine the role of soil to sequester C and to design management practices to maximise soil carbon storage. In addition, how to scale the plot scale modelling prediction to regional or continental scales is a big challenge (Saby et al., 2008; Ogle et al., 2009). For spatial modelling purposes, the uncertainty related to spatial variability of soil properties and the complexity of land use systems and management histories have to be considered. Different models may need to be combined in order to facilitate large scale predictions and enhance process level understanding.

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