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The legacy of cropping history reduces the recovery of soil carbon and nitrogen after conversion from continuous cropping to permanent pasture

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ABSTRACT

Enhancing soil organic carbon (SOC) and total nitrogen (N) is considered an important step in mitigating greenhouse gas emissions and improving soil fertility. The loss of SOC and N generally observed during cropping may be reversed by converting such land use to permanent pasture. However, large uncertainties remain around the processes that govern how much C and N may be sequestered from this conversion in soils worldwide. Here, we sampled soils across 10 paddocks on 20-year old grass pasture sites with a chronosequence of cropping history in order to quantify changes in SOC and N after the conversion of long-term cropping to pasture land use in a semi-arid region of southwest Queensland, Australia. The average rate of change in SOC stocks under pasture in the top 0–0.1 m soil layer was approximately 0.1 t C ha^{-1} yr^{-1}, consisting of an increase in SOC_{C4} (pasture) of 0.2 t C ha^{-1} yr^{-1} and a decrease in SOC_{C3} (pre-pasture) of 0.1 t C ha^{-1} yr^{-1}. The decrease in SOC_{C3} was enhanced at sites with greater years under cropping, indicating a reduced potential for SOC sequestration at sites with longer duration under cropping. The loss of total nitrogen (N) under cropping was also partially recovered with the introduction of permanent perennial pastures. A significant, positive correlation between soil aggregation and mineralisable N under cropping suggested that soil structure has a strong influence over N stability in the soil. However, soil aggregation and mineralisable N did not improve under pastures, indicating that the loss of soil fertility and structure under cropping remained a residual effect that was not recovered within 20 years of permanent pastures in this semi-arid subtropical environment. We suggest here that the resilience of ecosystems to recover soil fertility under pastures declines with greater years under cropping.

1. Introduction

Many soils have lost soil organic carbon (SOC), total nitrogen (N) and structure due to the conversion of remnant vegetated land to agricultural land use, contributing to increasing atmospheric greenhouse gas emissions as well as declining soil fertility and quality (Lal, 2004, 1993). The magnitude of the impact of land use conversion differs depending on the ecosystem's resistance to change and the type of agricultural activity undertaken post-conversion (Schipper et al., 2010; Seybold et al., 1999). This impact is particularly high in intensive agricultural activities such as cropping since the change can cause significant disturbance to the soil, leading to the loss of the natural soil structure that once stabilised soil organic matter (SOM), which contains both SOC and N (Jastrow et al., 1996). After conversion to cropping, the higher output of carbon and nitrogen from the system without sufficient input to counter-balance will therefore lead to net loss from the system (Fearnside and Barbosa, 1998). Approaches are sought to recover the loss of soil fertility by either increasing the inflow or reducing the outflow of SOM from the soil.

The impact of cropping on SOC and total N stocks has been studied extensively in previous studies, with long-term cropping sites indicating a steady loss of SOM with cropping age (Poeplau et al., 2011). This is largely due to the loss of C-stabilisation mechanisms provided by soil aggregates (Elliott, 1986; Six et al., 2000) which are reduced by tillage activities enhancing SOM decomposition during cropping (Dalal et al., 1991) and reduced C inputs. Cropping systems in Australia typically take at least 10–20

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years after disturbance to reach a new state of equilibrium (or steady-state) where the SOC and total N inflow equals outflow and may often take longer depending on soil type (Dalal and Mayer, 1986a; Luo et al., 2010). The decline in SOC and total N has considerable follow-on effects on soil fertility, including nutrient supply, particularly affecting the land suitability for further cropping and plant production (Dalal and Mayer, 1986a,b,c). The loss of soil aggregation also adversely affects water storage and infiltration, nutrient storage and micro-habitats for soil biota and increases susceptibility to erosion. To mitigate such losses in SOC and soil fertility—which have been identified worldwide (Guo and Gifford, 2002)—it is necessary to investigate the potential benefits of improved management practices that may recover these measures of soil quality and fertility while maintaining sufficient monetary advantage for landholders. The conversion to permanent pasture land-use may provide one such alternative in Australia (Glover et al., 2010), but questions about such benefits worldwide remain largely unanswered.

Many studies have demonstrated that the introduction of pastures for grazing activities after cultivation may recover soil structure due to increases in SOM, roots and rhizosphere activity (Carter and Stewart, 1995; Elliott, 1986), and act as a buffer to SOC loss after clearing, or may even restore organic carbon in soil after a substantial period of cultivation (Conant et al., 2001; Guo and Gifford, 2002; Poeplau et al., 2011; Sanderman et al., 2013). Gains in SOC after the conversion from cultivation to pasture on the global scale have been estimated to be as high as 19% (Guo and Gifford, 2002) and governed primarily by pasture age (Poeplau et al., 2011) and the stability of the existing C in the soil. However, there have been limited investigations into how the legacy of previous cropping activity may impact the resilience of agricultural ecosystems to recover lost soil structure, SOC and N (Seybold et al., 1999; Smith, 2014).

The question of ecosystem resilience to recover soil fertility and sequester SOC after conversion from cropping to pasture land-use requires revisiting as recent studies suggest that such gains under pastures may not be as clearly dependant on pasture age as originally thought (Smith, 2014). The "legacy" of prior cropping activity is a factor that is likely to impact on the recovery of soil aggregation and N availability, and influencing SOC sequestration potential. For example, land with a long history of cropping activity may contain high nutrient loads from residual fertilisation treatment for decades and even centuries (Smith, 2014), retain major disturbance of the natural soil structure during tillage (Carter and Stewart, 1995) or continue depletion of nutrients through crop produce removal (Dalal and Mayer, 1986b,c). Such cropping effects may remain as a legacy in the system and influence the biomass productivity or rate at which soils reach a new steady state after the conversion to a new land-use, but few studies have sought to answer these questions.

This study had unprecedented access to archived soil samples of cropping sites during the 1960s to 1980s (see Dalal and Mayer, 1986c) that have since undergone conversion to pasture land-use for similar periods of time. By using both archived (cropping) and new (pasture) samples, the changes in soil fertility and SOC after virgin-to-cropping then cropping-to-pasture land use conversions were measured with accuracy and confidence that is rarely available in such time-orientated studies. With the use of a series of cropping sites spanning 12–33 years since conversion from undisturbed natural vegetation ("virgin") to cropping land-use (herein a "chronosequence"), this study investigated the capacity of the agricultural ecosystems to recover SOC and soil fertility after conversion to permanent pastures. Specifically, all sites have undergone twenty years of pasture land-use since cessation of cropping activities, hence differences in soil fertility and SOC gains between sites according to cropping history could be identified. Building on our previous report (Jones et al., 2014), we sought to answer the question 'how does prior cropping activity impact the resilience of soil ecosystems to recover soil organic matter, aggregation and fertility?'.

2. Materials and methods

2.1. Study area

This study was conducted on the red earth soil (Kandosol, Rhodic Paleustalfs) in southwest Queensland, Australia (26.90 °S, 148.89 °E). The semi-arid climate of this area has a mean annual rainfall of 583 mm with temperatures ranging from 6 °C in winter to 36 °C in summer. The red earth soils of this region occur on a gently undulating landscape (slope <1%) as deep, dark reddish brown, sandy clay loams (approximately 23–37% clay content) with pH ranging from 6.4 to 7.0 (Table 1) and occasional calcareous horizons at depths >0.3 m. The soil clay mineralogy is made up of kaolinite (dominant), illite (accessory) and hematite (trace) with small amounts of iron oxides, originating from a parent material of weathered ferruginized sediments (Dalal and Mayer, 1986c). The original remnant vegetation (virgin sites), which exists as patches on landholder's paddocks, is open to tall shrublands, open woodlands and woodlands dominated by C3-vegetation; Eucalyptus populnea (Poplar Box), E. melanophloia (Silver-leaf ironbark) and Ermophila mitchellii (False sandalwood).

Continuous cereal cropping was the initial main land use for the area since the mid 1900s but the decline in soil fertility (a consequence of long-term unsustainable cropping practices) has resulted in an area-wide transition to grazing land use (Dalal and Mayer, 1986c). It is assumed that, prior to conversion to cropping, all sites within the same farm have the same soil properties as the adjacent native vegetation state.

2.2. Field site descriptions

Ten paddocks on two farms, Rockies and Rolston, that were previously selected in (Dalal and Mayer, 1986a,b,c) study to span several years under cropping and have since converted to grazing pasture were re-sampled in 2013 (Table 2). All 10 paddocks had been under similar grazing pastures for similar duration (19 or 20 years) so the differences in time previously under cropping may be used comparatively between paddocks. After conversion from continuous C3-cereal cropping, the paddocks were sown with C4-buffel grass (*Cenchrus ciliaris* L.). Information on the paddocks histories during cropping was available from previous studies (Dalal and Mayer, 1986c), and more recent management history was retrieved from current landholders. These paddocks had no prior fertilisation or N-fixing legumes either under cropping or pasture.

Table 1
Mean EC, pH, silt and clay contents of the sites at two farms under pastures. Standard error given in parenthesis.

Farm	Depth (m)	*n*	EC (dS m^{-1})	pH	Silt (%)	Clay (%)
Rockies						
	0–0.1	6	0.03 (±0.002)	6.7 (±0.2)	10 (±0.3)	26 (±1)
	0.1–0.2	6	0.02 (±0.002)	6.5 (±0.1)	10 (±0.6)	30 (±1)
	0.2–0.3	6	0.03 (±0.014)	6.4 (±0.1)	10 (±0.5)	32 (±1)
	0.3–0.6	6	0.10 (±0.029)	6.7 (±0.2)	10 (±0.5)	37 (±1)
Rolston						
	0–0.1	4	0.03 (±0.01)	6.5 (±0.1)	10 (±2.6)	23 (±1)
	0.1–0.2	4	0.03 (±0.01)	6.6 (±0.2)	10 (±1.4)	27 (±3)
	0.2–0.3	4	0.09 (±0.07)	6.9 (±0.5)	9 (±1.0)	32 (±6)
	0.3–0.6	4	0.18 (±0.16)	7.0 (±0.6)	9 (±1.6)	37 (±5)