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Long-term land use change in Australia from native forest decreases all fractions of soil organic carbon, including resistant organic carbon, for cropping but not sown pasture

Ram C. Dalal [a], Craig M. Thornton [b], Diane E. Allen [c], Jo S. Owens [c,d], Peter M. Kopittke [a,*]

[a] *The University of Queensland, School of Agriculture and Food Sciences, St Lucia, Queensland, 4072, Australia*
[b] *Department of Natural Resources, Mines and Energy, Rockhampton, Queensland, 4700, Australia*
[c] *Department of Environment and Science, Brisbane, Queensland, 4001, Australia*
[d] *University of Southern Queensland, Centre for Agricultural Engineering, Toowoomba, Queensland, 4340, Australia*

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ABSTRACT

Soil organic matter (SOM) performs an essential function in soil fertility, biomass and crop productivity, environmental sustainability, and climate change mitigation. We examined how land use change from native forest to either pasture [sown buffel (*Cenchrus ciliaris* cv. Biloela)] or cropping [primarily wheat (*Triticum aestivum* L.) and sorghum (*Sorghum bicolor* L.)] affected total soil organic C (SOC) stocks as well as stocks of three SOC fractions, particulate organic C, humus organic C and resistant organic C. Furthermore, for the cropping system, we also examined whether the use of a ley pasture phase could reverse the loss of SOC. It was found that land use change from native forest to pasture decreased SOC stocks by 12.2 % and soil total N (STN) stocks by 24.6 % during the land development to pasture establishment ($\leq$ 1.75 y), although there were no significant ($P > 0.05$) changes thereafter up to 33 y and final values were generally similar to initial values. Furthermore, stocks of the three SOC fractions did not change with time in this pasture system. In contrast to these modest changes following conversion to pasture, for land use change to cropping, SOC decreased by 48 % at 0–0.1 m and 38 % (from 54 to 33 Mg ha^{-1}) at 0–0.3 m, due mainly to insufficient C inputs to maintain SOM at steady state. Moreover, stocks of all three SOC fractions decreased with time, including the resistant organic C fraction, indicating that this fraction was not recalcitrant under cropping. The biomass C inputs by crops, mainly as root biomass, were not sufficient to reverse or slow down the rate of decrease of SOC in this soil. However, the introduction of pasture during the last 4 y indicated that the decreases in the stocks of SOC could be arrested by a ley pasture phase.

1. Introduction

Organic C and N are the integral components of soil organic matter (SOM), which is essential for agricultural sustainability, terrestrial environmental stability, and provides a long-term terrestrial C sink (Chenu et al., 2019). However, changes in land use can markedly alter stocks of soil organic C (SOC), with the magnitude of this change depending upon a broad range of factors, including the nature of the final land use. Firstly, consider a change in land use from native forest to introduced pasture, with this generally shown to cause only a comparatively modest change in SOC stocks. A global meta-analysis found a median decrease in SOC stocks of 11.3 % (Kopittke et al., 2017), with a similar value also reported in the meta-analysis of Don et al. (2011). Indeed, a change in land use to pasture can potentially increase, decrease or cause no change in SOC stocks (Murty et al., 2002; Kopittke et al., 2017), depending upon many factors, including soil type (Schipper et al., 2010). For example, Harms et al. (2005) studied the effect of land use change from native vegetation to pasture at 32 paired-sites in southern and central Queensland, Australia. Although SOC stocks decreased by approximately 7% across all sites, significant ($P < 0.05$) decreases in SOC stocks were found mostly in coarse-textured soils but not finer-textured soils.

In contrast to a change in land use to pasture, land use change from native vegetation to arable cropping generally leads to much larger losses of SOC and soil total N (STN). In the meta-analysis of Kopittke et al. (2017), the median decrease in SOC stocks upon conversion of land

* Corresponding author.
E-mail addresses: r.dalal@uq.edu.au (R.C. Dalal), Craig.Thornton@dnrme.qld.gov.au (C.M. Thornton), diane.allen@des.qld.gov.au (D.E. Allen), jo.owens@usq.edu.au (J.S. Owens), p.kopittke@uq.edu.au (P.M. Kopittke).

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from native vegetation to cropping was 46 %, with Don et al. (2011) reporting a value of ca. 20–35 %. For example, Dalal and Mayer (1986a) and Dalal and Mayer (1986b) found for a series of soils in Queensland (Australia) that SOC stocks decreased by 40–65 % and that STN stocks decreased by 40–57 % at 0−0.1 m depth in a range of Vertosols (Vertisols) and Kandosols (Ultisols). Corresponding stock losses at 0−0.3 m depths were 20–49 % for SOC and 25–45 % for STN, resulting in SOC decreases of 9.6-23.0 Mg ha^{-1} and STN decreases of 0.7-1.9 Mg ha^{-1}. In a similar manner, in a study from Missouri (USA), Motavalli and Miles (2002) found that conversion of native prairie to cropping reduced SOC stocks by up to ca. 70 %. Understanding such changes in SOC stocks in agricultural soils is critical given that food production must increase by 70 % between 2005 and 2050 (ELD, 2015).

These larger decreases in SOC stocks upon conversion to cropping are due to multiple factors. Firstly, it is noteworthy that Dalal and Mayer (1986a) and Dalal and Mayer (1986b) found that the rates of decrease in SOC in these soils were positively related to enzyme activity (urease) and negatively related to aggregation (aggregation index), thereby suggesting that the mechanism of microbial enzyme-accessibility to SOC was responsible for its turnover, which was later also proposed by Dungait et al. (2012) and Liang et al. (2017). This may explain, at least partly, the much larger decrease of SOC under cropping than pasture; in the former the microbial enzyme-accessibility to SOC is substantially increased due to soil disturbance from tillage operations. In addition, lower root biomass C inputs in the soil under cropping than pasture further leads to decreases in SOC stocks with time (Dalal et al., 2005a, 2013; Beniston et al., 2014). However, the introduction of a legume-based ley pasture phase into cropping rotation arrests SOC decline and may even increase SOC stocks by increasing total stock of root C, reducing N loss in grain harvest as well as N addition by pasture legume (Dalal et al., 1995; Beniston et al., 2014; Thornton and Shrestha, 2020).

Until this point, we have only considered bulk concentrations of organic C, but land use change may affect SOC fractions differently. For example, Baldock et al. (2013a) and Baldock et al. (2013b) suggested that among the SOC fractions; particulate organic C (POC), mineral-associated organic C (MAOC) and resistant organic C (ROC), the ROC fraction was biologically least reactive and may not change under different land uses, including cropping. In fact, SOC stocks under cropping, using the models, RothC (Skjemstad et al., 2004) and APSIM (Luo et al., 2014), were successfully simulated by assuming ROC as the resistant / inert SOC pool. However, there is a paucity of data globally examining temporally-measured changes in the SOC fractions; POC, MAOC and ROC, over a long period under either pasture or cropping land use to unequivocally confirm that the ROC fraction is indeed an inert SOC fraction.

The long-term monitoring of the Brigalow Catchment Study (1965–2014) (Cowie et al., 2007; Thornton and Shrestha, 2020) in Queensland (Australia), consisting of pasture and cropping sites and adjacent native forest, provides an opportunity to examine whether land use change from native forest to either pasture or cropping in a low input system not only causes differences in the SOC and STN stocks over a 33 y period, but also whether it concomitantly changes the SOC fractions. This monitoring site is located in a semi-arid to subtropical system. We measured the SOC and STN stocks, and the SOC fractions at 0−0.1 m, 0.1−0.2 m, 0.2−0.3 m, and 0.3−0.4 m depths from 1981 to 2014 at different periods following land use change from the native forest to either pasture or cropping. Changes in the SOC source, whether derived from the C_3-native forest, C_4-pasture or C_4-crops were measured using δ^{13}C values ($C_{3vegetation}$, from -22‰ to -30‰; $C_{4vegetation}$, from -10‰ to -16‰, Balesdent et al. (1987)) and N transformation of STN using δ^{15}N values (Robinson, 2001) from 1981 to 2014. The objective of this study was to examine the temporal effects of land use change on C and N stocks, together with the different fractions of SOC, in a semi-arid to subtropical soil in Australia. To do this, we: i) quantified the stocks of SOC and STN; ii) estimated the SOC fractions: POC, MAOC and ROC, using mid infra-red (MIR) spectroscopy; and, iii) measured the δ^{13}C values of SOC and δ^{15}N values of STN at 0−0.1 m, 0.1−0.2 m, 0.2−0.3 m, and 0.3−0.4 m depths, following land use change from the native forest to either pasture or cropping over a 33 y period, at the Brigalow Catchment Study, Queensland, Australia.

2. Materials and methods

2.1. Sampling site

The study site is located at 24.81 °S, 149.80 °E at an altitude of 151 m above sea level. Mean annual rainfall is 720 mm and annual potential evaporation is 2100 mm. The mean maximum temperature is 33.1 °C in January and minimum temperature is 6.5 °C in July (Cowie et al., 2007; Thornton and Shrestha, 2020).

The main soil types at the site are Grey Vertosols/Dermosols and Sodosols (Isbell, 2002), Vertisols (IUSS Working Group WRB, 2015), or Ustic Pelluserts (Soil Survey Staff, 2014). Clay content of the dominating soil type, Vertisol (Vertosol), was 36 % and pH 6.6 at 0−0.1 m depths across the site (Cowie et al., 2007). The dominant native vegetation at the site is brigalow (*Acacia harpophylla*). Belah (*Casuarina cristata*) and blackbutt (*Eucalyptus cambageana*) are the co-dominant species. A detailed description of the site is available from Cowie et al. (2007).

The experimental site consists of three adjacent sites (each being 12−17 ha in size) with similar area, slope, aspect, vegetation, and soils (Cowie et al., 2007). Each site formed a land use treatment, with the first site retained as a control (native forest), the second site converted to cropping, and the third site converted to pasture (see later). Between March and October 1982, the native forest on the second and third site was cleared for cropping and pasture respectively by pulling, burning and raking. The pasture site was sown to buffel pasture (*Cenchrus ciliaris* cv. Biloela) in November 1982. Grazing of the pasture commenced a year later and the animal stocking rate was adjusted to maintain ground cover >85 % or about 1000 kg ha^{-1} of aboveground pasture biomass during the experimental period (1983–2014) (Radford et al., 2007; Thornton and Shrestha, 2020). Estimations of annual pasture biomass growth for the pasture site were determined using the GRASP pasture model (Rickert et al., 2000). Data from the Brigalow Catchment Study has been used by GRASP modellers for many decades, with the outputs contributing to Queensland wide estimations of pasture growth through FORAGE and Australia wide estimations of pasture through the AussieGRASS model (Carter et al., 2000), available via the Long Paddock website (The State of Queensland, 2020). The GRASP model was run with site measured and estimated parameters of soil and vegetation for the pasture site for the period 1890–2020, using stocking rates from the start of grazing in 1983. The model was calibrated to measured pasture yield data from 1983 to 2018 using the Botanal method (Radford et al., 2007). This site produced 238 Mg ha^{-1} of aboveground biomass (C_4-pasture), with 49 kg ha^{-1} of total N removed in animal product (beef) over a 32 y pasture growth (Thornton and Shrestha, 2020).

For the cropping site, grain sorghum (*Sorghum bicolor* L.) was sown once in September 1984 (summer), 2.5 y after pulling and 1.9 y after clearing. Thereafter, for Phase 1 of cropping, wheat (*Triticum aestivum* L.) was grown annually (winter) for 9 y except for a drought year in 1993. During this period of wheat monoculture, weeds during the fallow periods were controlled by conventional tillage (disc plough, chisel plough, and scarifier). From 1995 (i.e. approximately 12 y from clearing, and after 10 y of Phase 1 cropping) until 2010, minimum tillage and opportunity cropping were practised. Thus, during this Phase 2 of cropping, the summer and winter crops, wheat, barley (*Hordeum vulgare* L.), chickpea (*Cicer aeritenum* L.) or sorghum, were sown whenever soil water content was considered adequate. During this time for Phase 2 of cropping, a total of 19 crops were harvested, being 10 sorghum, four wheat, one barley, and one chickpea. No fertiliser was applied since fertility decline could be monitored without the confounding effect of fertiliser input. In 2010 (i.e. after 28 y total), butterfly pea (*Clitoria ternatea*) was sown as a ley pasture, a means of restoring soil fertility. It was grazed 15 months later and only lightly grazed since then, about 0.3–0.5 equivalent stocks ha^{-1}, with regular periods of pasture spelling. Over the 26-y period of arable cropping (1984–2010), this site produced 11 sorghum crops with a total grain yield of 20.5 Mg ha^{-1}, 13 wheat crops (27.5 Mg ha^{-1}), one barley crop (0.6 Mg ha^{-1}) and one chickpea crop (0.5 Mg ha^{-1}), that is, a total grain yield of 49 Mg ha^{-1}, and total N removal in the grain was 0.958 Mg ha^{-1} (Thornton and Shrestha, 2020). The crop residues were retained *in situ* on the site.

2.2. Soil sampling and soil organic matter fractionation

Soil samples were collected in 1981 (baseline, 0 y), 1983, 1985, 1987, 1990, 1994, 1997, 2000, 2008 and 2014 at three monitoring sites within each of the three treatments (sites) of native forest, pasture and cropping. At each monitoring site (20 m × 20 m), five samples were taken down to 0.4 m depths by a hydraulic-driven corer (0.05 m diameter) and divided into 0–0.1 m, 0.1–0.2 m, 0.2–0.3 m, and 0.3–0.4 m depths, and bulked for respective depths. These bulk samples from the three monitoring sites within a site were then bulked to create a representative sample for each site and depth increment. More details on the sample sites within each treatment (site) are available from Cowie et al. (2007).

Soil samples were dried at 40 °C and ground to pass <2 mm sieve prior to analysis. Intact soil cores were also used for bulk density measurements. Sub-samples were dried at 105 °C to a constant weight and bulk density was calculated from the total sample mass, corrected to oven dry mass of soil and internal core volume.

These measurements of bulk density were necessary for correction for equivalent soil mass to that under the native forest. The bulk density of the forest soil did not change significantly ($P > 0.05$) although that of the pasture soil and cropped soil increased at 0–0.1 m depths during the monitoring period. Specifically, the average bulk density (0–0.1 m depths) over the entire monitoring period was 1.2 Mg m^{-3}, whilst over the 33 y monitoring period the bulk density increased from 1.1 to 1.5 Mg m^{-3} for pasture and from 1.1 to 1.4 Mg m^{-3} for cropping (mean values across the 33 y monitoring period are presented in Table S1). Below these depths, bulk densities of the soil under all three land uses followed a similar trend: cropping > pasture > native forest. The cumulative amounts of SOC and STN (see later) for 0–0.1 m, 0–0.2 m, 0–0.3 m, and 0–0.4 m depths were, therefore, corrected for equivalent soil mass to that under the native forest, using the polynomial relationship (r^2 >0.99) between the amount of SOC and soil mass (soil depth × bulk density) (Dalal et al., 2005a, b). The equivalent soil mass values used for all three land uses for each depth interval were 1101, 2287, 3682 and 5136 Mg ha^{-1} for 0–0.1 m, 0–0.2 m, 0–0.3 m and 0–0.4 m, respectively. Soil depths reported hereafter, therefore, refer to those of the native forest. Similar calculations were made for the STN, the SOC fractions, and $\delta^{13}C$ and $\delta^{15}N$ values (see later).

The organic C fractionation procedure has been described previously (Baldock et al., 2013a, b). The soil fractions, POC, MAOC, and ROC were calculated using MIR spectroscopy of the air-dried soil samples (<0.25 mm size), which was previously calibrated from 312 standard samples. The MIR spectra were analysed using partial least-square regression. The POC comprised of SOC > 0.05 mm, free from resistant organic C including char C, the MAOC comprised of SOC < 0.05 mm, free from resistant organic C including char C, and the remaining fraction was ROC, which was estimated from the above two fractions using ^{31}C NMR spectrometry (Baldock et al., 2013b).

2.3. Soil analysis for organic carbon, N and $\delta^{13}C$ and $\delta^{15}N$

Total SOC and natural abundance ^{13}C of the whole samples (ground to <0.1 mm) were determined using an Isoprime isotope ratio mass spectrometer (IRMS) coupled to a Eurovector elemental analyser (Isoprime-EuroEA 3000, Milan, Italy). Samples containing inorganic carbonates were pre-treated with H_2SO_3 before analysis. SOC was measured by dry combustion method (LECO Corporation, Michigan, USA). The isotope ratios were expressed using the 'delta' notation (δ), with units of per mil or parts per thousand (‰), relative to the marine limestone fossil Pee Dee Belemnite standard (Craig, 1953) for $\delta^{13}C$ (Eq. 1):

$$\delta^{13}C\ (‰) = (R_{sample} / R_{standard} - 1) \times 1000 \quad (1)$$

where R is the molar ratio of $^{13}C/^{12}C$ of the sample or standard (Ehleringer et al., 2000).

The proportion of organic C in soil derived from C_4 vegetation, was estimated by using a mixing model (Dalal et al., 2005a), (Eq. 2):

$$\text{Soil } C_4\text{-derived C } (C_4\text{-SOC}) = (\delta^{13}C_{soil\ under\ pasture} - \delta^{13}C_{C3\ soil\ under\ native\ forest}) / (\delta^{13}C_{C4pasture} - \delta^{13}C_{C3\ soil\ under\ native\ forest}) \quad (2)$$

where $\delta^{13}C_{soil\ under\ pasture}$ is the $\delta^{13}C$ value of soil organic C under pasture, $\delta^{13}C_{C4}$ is the average value of $\delta^{13}C$ value of C_4 buffel pasture ($\delta^{13}C_{C4pasture}$ = -14.63‰) and $\delta^{13}C_{C3\ soil\ under\ native\ forest}$ is the $\delta^{13}C$ value of SOC under native forest (-23.0‰). A similar equation was used to estimate the proportion of SOC derived from C_4 sorghum crops ($\delta^{13}C_{C4sorghum}$ = -12.5‰, Kurdali (2009)) in the cropping soil.

The proportion of previous forest C_3-derived C (C_3-SOC) in the soil under pasture was the difference between the total SOC and the amount of C_4-SOC in the pasture soil. Similar calculations were made at individual depths and in different fractions for the C_4-SOC in the pasture and the cropped soil, using the corresponding $\delta^{13}C$ values. No corrections, therefore, were required for Rayleigh fractionation of enrichment of $\delta^{13}C$ values with soil depth. For the cropping soil, whilst the C_4-SOC derived from C_4-crops (sorghum) can be determined, it is not possible to separate the C_3-SOC derived from the C_3-native forest and the C_3-SOC derived from the C_3-crops.

Natural abundance ^{15}N of soil samples from the 0–0.1, 0.1–0.2, 0.2–0.3 m, and 0.3–0.4 m depths were determined using an Isoprime isotope ratio mass spectrometer coupled to a Eurovector elemental analyser (Isoprime-EuroEA 3000, Milan, Italy). Samples containing approximately 50 μg N, estimated from STN which was measured using the dry combustion method (LECO Corporation, Michigan, USA), were weighed into 8 × 5 mm tin (Sn) capsules and analysed against a known set of standards. The isotope ratios were expressed using the 'delta' notation (δ), with parts per thousand (‰), relative to N_{air} standards for $\delta^{15}N$ using the following relationship:

$$\delta^{15}N\ (‰) = (R_{sample} / R_{standard} - 1) \times 1000 \quad (3)$$

where R is the molar ratio of the heavy to light isotope (i.e. $^{15}N/^{14}N$) of the sample or standard.

2.4. Turnover of carbon in whole soil and soil fractions

The rates of loss of native forest C_3-derived C in the whole soil and soil fractions were calculated for the pasture and cropping soils, as follows (Dalal and Mayer, 1986a):

$$C_t = C_e + (C_o - C_e)\exp(-kt) \quad (4)$$

where C_o (Mg ha^{-1}) is the amount of SOC initially under the native forest, pasture or cropping at 0 y and C_t (Mg ha^{-1}) are the amounts of SOC at different periods after clearing of the native forest under pasture or cropping, C_e (Mg ha^{-1}) is the amount of SOC at steady state, t is the monitoring period (year), and k (1/year) is the rate of loss of SOC from soil. A similar equation was used for the changes in C_3-SOC and C_4-SOC under pasture or cropping during the monitoring period. The turnover period of SOC under pasture or cropping was the reciprocal of k (1/k, year). Eq. 4 was also used to describe the changes in stocks of the SOC fractions and STN, and $\delta^{13}C$ and $\delta^{15}N$ values, with the periods of monitoring at different depths.

2.5. Statistical analysis

A three-way analysis of variance (ANOVA) of land use, soil depth and year of monitoring and their interactions on the SOC and the SOC fractions, STN and δ^{13}C and δ^{15}N was conducted (Table S2) (R Core Team, 2017). The structure of ANOVA was: land use x depth x time + error (replication).The results are presented as mean values with standard errors. The values of C_o, C_e, k and 1/k and their standard errors were obtained from Eq. 4 using SigmaPlot 14.0 (SYSTAT Software). When the exponential coefficient was not significant ($P > 0.05$), a simple linear regression between C_t and t was applied. Other variables were processed similarly.

3. Results

3.1. Overall nature of changes in SOC and STN

From the three-way ANOVA, it was found that there was a significant (P < 0.05) interaction between land use, depth, and time, with this being for both SOC and STN (Table S2). ANOVA outputs of SOC fractions and δ^{13}C and δ^{15}N are also reported in Table S2. The magnitude of change depended upon land use, with the change being substantially larger for cropping than for pasture (see later). The pattern of change in SOC and STN depended upon both the depth (changes tending to be greater near the soil surface) and time (changes tending to increase over time).

To provide a more detailed assessment of how land use (pasture and cropping) affected SOC and STN stocks over time, we used regression analyses to determine the depth to which SOC and STN changed. For cropping, it was found that SOC stocks within the individual depth increments decreased significantly (P < 0.05) for the 0−0.1, 0.1−0.2, and 0.2−0.3 m increments, but that there were no significant (P > 0.05) changes for the 0.3−0.4 m increment (Table 1). Thus, we hereafter focus primarily on changes in the SOC stocks to a depth of 0.3 m for both pasture and cropping soils. Moreover, for C accounting purposes, IPCC (2006) recommends 0−0.3 m depths following land use change.

3.2. Pasture: Soil organic carbon and carbon distribution in different soil fractions

Except for the loss of 3.0 Mg ha^{-1} of SOC during the pasture development phase (clearing, burning, land preparation and establishment) within 1.75 y (24.2 ± 1.6 at 0 y versus 21.2 ± 1.6 Mg ha^{-1} at 1.75 y), the total SOC stocks at 0−0.1 m depth did not change during the remaining monitoring period (Fig. 1a, b). Indeed, following this initial rapid loss of SOC, stocks remained relatively constant, perhaps increasing slightly (although not significantly (P > 0.05)), with the final value after 33 y (24.6 ± 2.4 Mg ha^{-1}) being similar to the initial value (24.2 ± 1.6 Mg ha^{-1}). Thus, although there was a modest rapid initial loss of SOC (3.0 Mg ha^{-1}), over the entire monitoring period, there was no significant (P > 0.05) change and the SOC stock after 33 y was similar to the initial value. This was also true for the 0−0.2 m and 0−0.3 m (Fig. 1a, also see Fig. S1a for each individual depth increment). Although total SOC stocks did not change over time, C_3-SOC derived originally from the native forest decreased exponentially from 24.2 ± 1.6 Mg ha^{-1} at 0 y to 9.0 ± 1.5 Mg ha^{-1} at steady state (0−0.1 m depth, Fig. 1b). The rate of C_3-SOC loss was 0.106 ± 0.033 y^{-1}, with a turnover time of 9.5 ± 2.9 y. Except for the C_4-pasture development phase, the loss of C_3-SOC was primarily compensated for by C inputs from the C_4 pasture (Fig. 1b).

The C_4-SOC under pasture increased from a value of -0.1 ± 4.6 Mg ha^{-1} at 0 y to 19.1 ± 5.4 Mg ha^{-1} at steady state (Fig. 1b). The rate of C_4-SOC increase was 0.046 ± 0.023 y^{-1} ($P = 0.08$), with a turnover time of 21.6±10.7 y. At 0−0.2 m depth, the C_3-SOC and C_4-SOC stocks changed linearly with time: the C_3-SOC stocks decreased by 0.58 ± 0.10 Mg ha^{-1} y^{-1} ($R^2 = 0.80$; $P < 0.01$) which was compensated for by an increase in C_4-SOC stocks by 0.58 ± 0.12 Mg ha^{-1} y^{-1} ($R^2 = 0.75$; $P < 0.01$). Below this depth, changes in the C_3-SOC and C_4-SOC stocks with time were not significant (P > 0.05) (data not presented). At the end of the monitoring period (33 y), average amounts of C_4-SOC stocks were 64 % of the total SOC stocks at 0−0.1 m, 51 % at 0−0.2 m, and 40 % at 0−0.3 m depths.

The stocks of the various SOC fractions (POC, MAOC and ROC) at 0−0.1 m remained similar in the soil under pasture throughout the monitoring period, with regression analyses indicating no significant (P > 0.05) changes over time (P > 0.05) (Fig. 1c). In a similar manner, below this depth, stocks of the various SOC fractions did not change during the monitoring period (see Fig. S1 for individual depth increments and Fig. S2 for cumulative depths). However, the relative contribution of POC to the total SOC stock increased with depth from 20.6 ± 0.6 % at 0−0.1 m to 25.4 ± 1.2 % at 0−0.3 m depths, whereas that of the MAOC decreased from 53.1 ± 0.6 % at 0−0.1 m depth to 48.2 ± 1.2 % at 0−0.3 m depths (Fig. S2). The proportions of ROC stocks (26.2 ± 1.0 %) remained essentially similar at all depths.

3.3. Pasture: Total nitrogen, δ^{15}N values and C:N ratio

Averaged across the entire monitoring period for soils converted to pasture, the stocks of STN were 1.71 ± 0.17 Mg ha^{-1} at 0−0.1 m, 2.80 ± 0.21 Mg ha^{-1} at 0−0.2 m, and 3.88 ± 0.26 Mg ha^{-1} at 0−0.3 m depths. Although STN stocks at 0 y were similar for both the forest (2.04 ± 0.13 Mg ha^{-1}) and pasture (2.12 ± 0.38 Mg ha^{-1}) sites, the STN stocks decreased rapidly during the land development for pasture establishment phase (<1.75 y, Fig. 2a), in a manner similar to that observed for SOC stocks. However, although STN stocks tended to increase slightly across the subsequent 33-y monitoring period, this change was not significant (P > 0.05) (Fig. 2a).

The δ^{15}N values of STN increased with depth; the average δ^{15}N values were 4.3 ± 0.5‰ at 0−0.1 m, 5.3 ± 0.7‰ at 0−0.2 m, and 6.0 ± 0.9‰ at 0−0.3 m (Fig. 2b). Similar to STN stocks, there was no significant (P > 0.05) change in δ^{15}N values at any depth over the monitoring period (Fig. 2b).

The C:N ratio decreased slightly with depth, with values being 13.5 ± 0.6 at 0−0.1 m, 13.1 ± 0.3 at 0−0.2 m, and 12.8 ± 0.2 at 0−0.3 m depths. This may be due to incorporation of biomass in the top layers in the early years. However, there was no significant (P > 0.05) change with time in C:N ratio at any depth during the monitoring period (Fig. S3).

3.4. Cropping: Soil organic carbon and carbon distribution in different soil fractions

At 0−0.1 m depth, SOC stocks under cropping decreased exponentially from 27.9 ± 0.6 Mg ha^{-1} at 0 y to 13.9 ± 0.6 Mg ha^{-1} at steady state, being a decrease of ca. 50 % (Fig. 3b and Table 2). Accordingly, the rate of loss of SOC was 0.119 ± 0.023 y^{-1}, with a turnover time of 8.4 ± 1.6 y at 0−0.1 m depth (Fig. 3b and Table 2). Furthermore, the rate of

Table 1
Discrete depth effect on stocks of soil organic C (SOC) and soil total N (STN) following land use change from native forest to cropping over 28 y. Land use change from native forest to pasture showed no significant (P > 0.05) trend during the monitoring period.

Depth (m)	Regression constant (Mg ha^{-1})	Regression coefficient (Mg ha^{-1} y^{-1})	R^2	Level of significance (P<)
SOC (Mg ha^{-1})				
0−0.1	26.0 ± 0.9	−0.029 ± 0.004^A	0.90	0.01
0.1−0.2	11.9 ± 0.5	−0.124 ± 0.039	0.59	0.05
0.2−0.3	11.5 ± 0.5	−0.108 ± 0.042	0.49	0.05
0.3−0.4	9.2 ± 0.5	−0.070 ± 0.042	0.29	ns
STN (Mg ha^{-1})				
0−0.1	2.35 ± 0.06	−0.037 ± 0.005	0.88	0.01
0.1−0.2	1.13 ± 0.04	−0.014 ± 0.003	0.71	0.05
0.2−0.3	1.11 ± 0.04	−0.010 ± 0.003	0.65	0.05
0.3−0.4	0.91 ± 0.03	−0.003 ± 0.002	0.23	ns

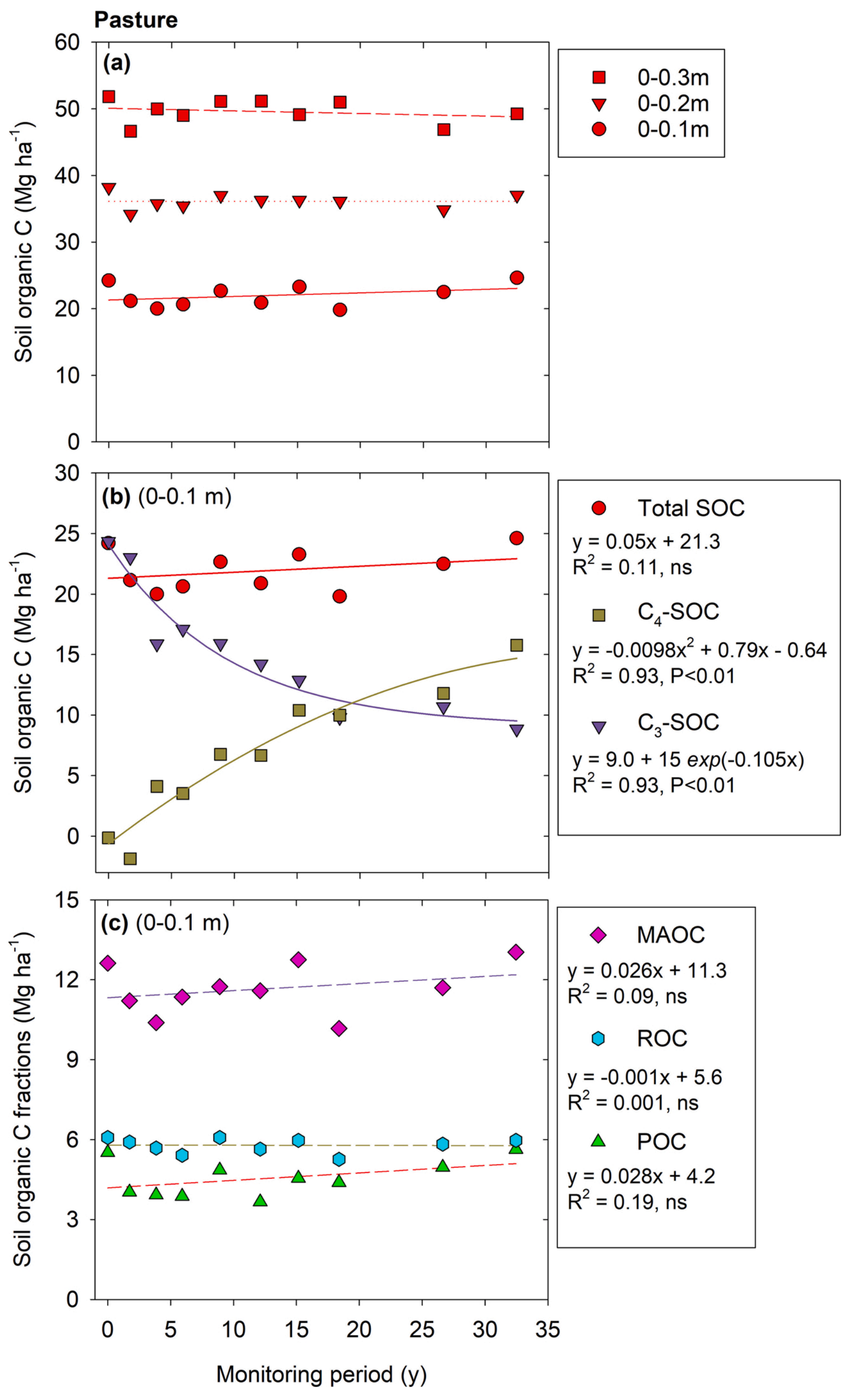

Fig. 1. Changes in (a) total soil organic C (SOC) stocks at 0-0.1, 0-0.2, and 0-0.3 m depths, (b) total soil organic C (SOC) stocks, C_3-SOC stocks (from native C_3 forest) and C_4-SOC stocks (from grass pasture) at 0-0.1 m depth, and (c) soil organic C fractions, particulate organic C (POC), mineral-associated organic C (MAOC) and resistant organic C (ROC) at 0-0.1 m depth, following land use change from native forest to pasture for 33 y (1981-2014).

SOC loss increased with increasing depth, with corresponding decreases in turnover times (Table 2 and Fig. 3a, and also see Fig. S4).

Total SOC under cropping was derived from the native forest (C_3-forest; original C_3-SOC) and the C_3 crops (wheat, barley, chickpea; crop C_3-SOC) as well as the C_4 sorghum crops (C_4-SOC). Over the 28 y period prior to the introduction of the ley pasture, the C_4-SOC stock increased linearly at a rate of 0.11 ± 0.022 Mg ha^{-1} y^{-1} (Fig. 3b), with the C_4-SOC stock after 28 y being 3.2 ± 0.6 Mg ha^{-1}. Simultaneously, there was a marked reduction in the C_3-SOC stock over time (Fig. 3b). Therefore, by the end of the monitoring period, C_4-SOC stock comprised 22 % of the total SOC at 0−0.1 m depth. At 0−0.2 m depth, the C_4-SOC increased at a rate of 0.168 ± 0.037 Mg ha^{-1} y^{-1}, (5.5 ± 1.2 Mg ha^{-1}), accounting for 22.6 % of the total SOC stock by the end of the monitoring period. However, no significant ($P > 0.05$) trends in C_4-stocks were observed below this depth.

All three SOC fractions (POC, MAOC, and ROC) decreased under cropping at 0−0.1 m depth (Fig. 3c and Table 3). However, the POC fraction decreased at almost twice the rate of the MAOC fraction, being 0.175 ± 0.021 y^{-1} for the POC fraction and 0.091 ± 0.047 y^{-1} for the MAOC fraction (Fig. 3c and Table 3). The rate of decrease of ROC fraction (0.089 ± 0.016 y^{-1}) was close to that of the MAOC fraction and not significantly ($P > 0.05$) different that from the whole SOC (0.119

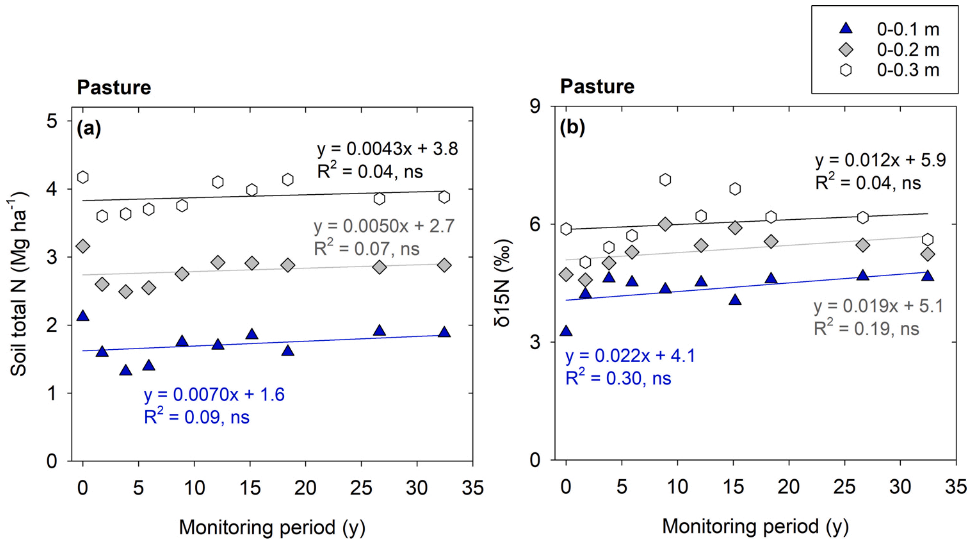

Fig. 2. Changes in soil total N (STN) stocks and δ^{15}N at 0-0.1 m, 0-0.2 m, and 0-0.3 m depths, following land use change from native forest to pasture for 33 y (1981-2014).

y^{-1}, Table 2). Initially the POC fraction was 25 % of the total SOC, however, at steady state it accounted for only 14 % of the total SOC. In contrast, the MAOC fraction increased from 47 % to 55 % of the total SOC whilst the ROC fraction remained essentially similar, being approximately 27–28 % of the total SOC stock. Similar trends were observed at 0−0.2 m and 0−0.3 m depths (Table 3 and Fig. S5), although only the MAOC and ROC fractions decreased exponentially with time, with the decrease in the POC fraction being linear across the period of monitoring (Table 3).

3.5. Cropping: Total nitrogen, δ^{15}N values and C:N ratio

Under cropping, STN stocks decreased linearly over time (Fig. 4 and Table 4). The STN stocks also decreased at 0−0.2 m and 0−0.3 m depths under cropping (Table 4). Specifically, the rates of decrease in STN stocks were 0.037 ± 0.005 Mg ha^{-1} y^{-1} at 0−0.1 m, 0.051 ± 0.007 Mg ha^{-1} y^{-1} at 0−0.2 m, and 0.061 ± 0.009 Mg ha^{-1} y^{-1} at 0−0.3 m depths (Fig. 4 and Table 4).

The δ^{15}N values of STN at 0−0.1 m increased with time from 5.9 ± 0.6–8.9 ± 0.8‰ over the first 28 y of the monitoring period (Fig. 4). However, after 28 y the legume ley pasture was introduced, with the δ^{15}N values of STN at 0−0.1 m then subsequently decreasing to 7.8‰ after 33 y (Fig. 4). A similar relationship was also observed at 0−0.2 m depth (Fig. 4).

Generally, the C:N ratio of the SOM tended to decrease up to 28 y, although after 33 y, the values were generally higher, resulting in a quadratic relationship, which was somewhat similar to the trends in δ^{15}N values although in the former this relationship was significant ($P < 0.05$) at all depths (Fig. S6).

4. Discussion

In the present study, land use change from native forest to pasture resulted in a rapid (albeit comparatively modest) decrease in SOC and STN stocks only during the land development to pasture establishment phase (≤1.75 y), resulting in a decrease of 12.6 % in SOC and 24.9 % in STN. However, across the remaining experimental period, stocks of SOC and STN did not differ significantly ($P > 0.05$) at the site. Similarly, stocks of the various SOC fractions (POC, MAOC and ROC) did not change with time upon conversion to pasture. It was also noted that although the δ^{15}N of STN did not change at any depth over time, the δ^{13}C of the SOC increased over time due to the C input by the C_4-pasture. In contrast to these observations for pasture, land use change to cropping led to large decreases in the SOC and STN stocks at all depths, due mainly to insufficient C inputs and N removal in the grain and runoff, leaching and gaseous N losses. Importantly, stocks of all three SOC fractions decreased with time, indicating that the ROC was not a recalcitrant fraction under cropping as might have been expected (Baldock et al., 2013a). The δ^{15}N values of STN were higher than those under native forest or pasture, indicating an increased transformation/ mineralisation of N (Ehleringer et al., 2000; Robinson, 2001), preferential ^{15}N crop N uptake and removal in grain harvest under cropping, compared to native forest or pasture sites. Finally, the introduction of the pasture phase in the crop rotation during the last 4 y of the monitoring period indicated that the decreases in the stocks of SOC and STN could be arrested by a ley pasture phase.

4.1. Soil organic C and soil total N dynamics following land use change from forest to pasture

Conversion of the native forest land to pasture at this site resulted in the loss of 3.0 Mg ha^{-1} of SOC, or a 12.6 % decrease during the first years (≤1.75 y) of land preparation for pasture establishment (Fig. 1b). The loss of STN during this same initial period was substantially larger (24.9 %), most likely via burning, resulting in heating of the soil surface and volatilisation, leaching and runoff (Cowie et al., 2007; Thornton et al., 2007; Silburn et al., 2009). For example, runoff from the cropping site increased by two-fold, whilst leaching increased to > 20 mm y^{-1} compared to <1 mm y^{-1} in the native forest (Cowie et al., 2007; Thornton et al., 2007; Silburn et al., 2009). Our findings that there was an initial, rapid decrease in SOC and STN are in accordance with previous observations. For example, in this bioregion, Pringle et al. (2016) found that SOC decreased by 25 % and that STN decreased by 19 % (0−0.3 m depth) following the clearing of native forest by burning and land development for pasture establishment at a number of sites. In our study, most of the STN stock losses (>90 %) during the development phase occurred at 0−0.1 m depth (0.527 Mg ha^{-1} at 0−0.1 m and 0.572 Mg ha^{-1} at 0−0.3 m, Fig. 2), thus supporting the hypothesis that the STN losses were due to erosion and runoff following burning of the biomass cover and soil disturbance for pasture establishment (Cowie et al., 2007; Hobley et al., 2017). Such losses of both the SOC and STN can be reduced by establishing pastures with minimum soil disturbance and retaining the biomass soil cover [for example, see the meta-analyses of Guo and Gifford (2002) and Kopittke et al. (2017)].

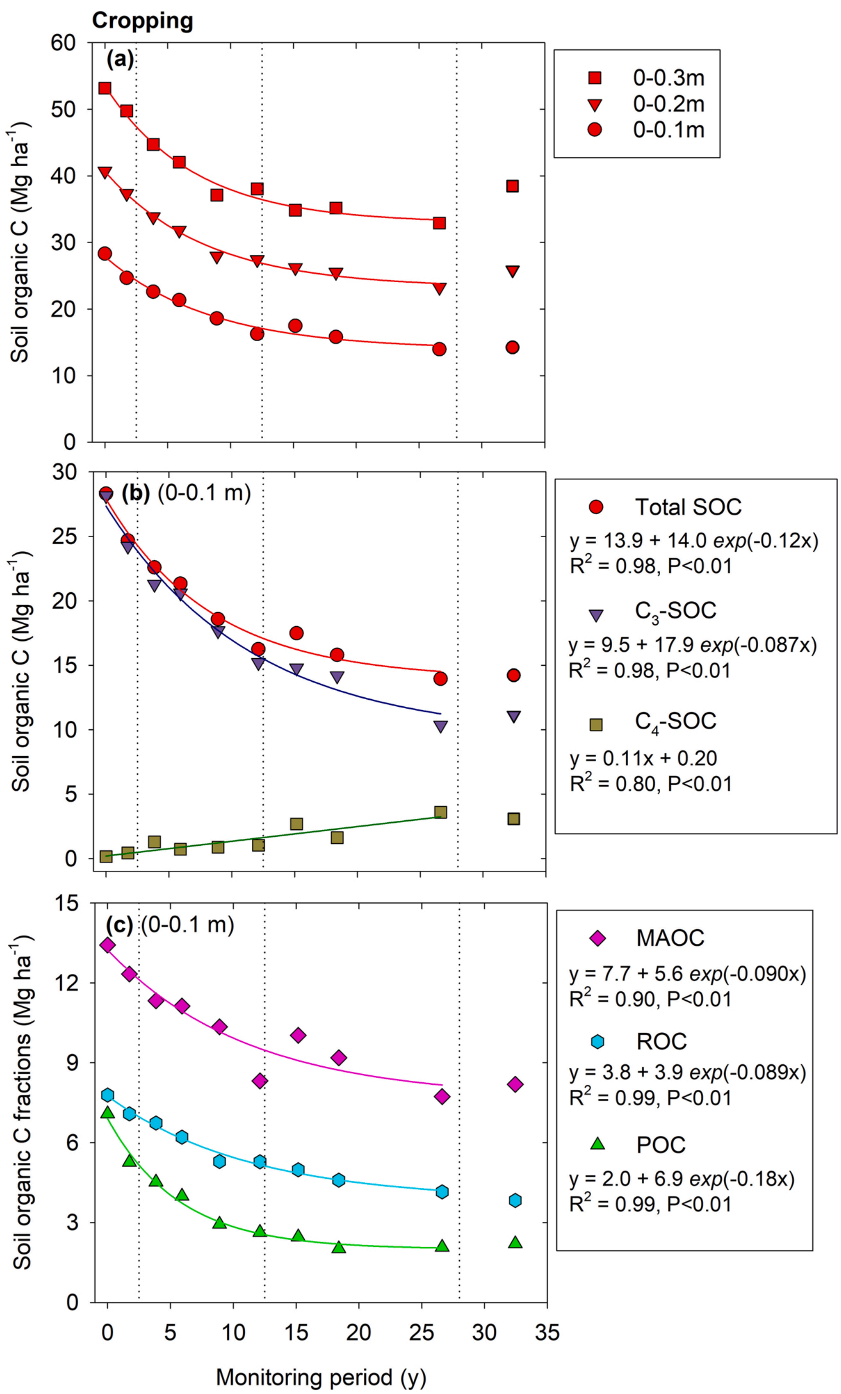

Fig. 3. Changes in (a) total soil organic C (SOC) stocks at 0-0.1, 0-0.2, and 0-0.3 m depths, (b) total soil organic C (SOC) stocks, C_3-SOC stocks (from native C_3 forest and C_3 crops) and C_4-SOC stocks (from C_4 crops) at 0-0.1 m depth, and (c) changes in soil organic C fractions, particulate organic C (POC), mineral-associated organic C (MAOC) and resistant organic C (ROC) at 0-0.1 m depth, following land use change from native forest to cropping for 33 y (1981-2014). The first crop (sorghum) was planted 2.5 y after pulling and clearing commenced. Thereafter, Phase 1 of cropping was a wheat monoculture. Phase 2 of cropping started 12 y after pulling and clearing commenced, with 10 sorghum crops, four wheat, one barley, and one chickpea. After 28 y, a ley pasture was sown (excluded from regression analysis).

After this initial loss of SOC and STN following land use change from the native forest to pasture, there was no significant ($P > 0.05$) change with time during the remaining 33 y monitoring period (Fig. 1b), thus corroborating the earlier findings of Radford et al. (2007) and Dalal et al. (2013). These findings are in general agreement with global meta-analyses, which have found that conversion of land to pasture results in only a comparatively modest changes in SOC stocks, often in the order of 10 % (Don et al., 2011; Kopittke et al., 2017). For example, Harms et al. (2005) reported <10 % decrease in SOC in Vertosols (Vertisols) following land clearing from forest and woodland to pasture in Central Queensland, Australia. Indeed, Murty et al. (2002) reported minor changes in SOC and STN stocks when land use change from forest to pasture occurred, especially in situations where grazing pressure or stocking intensity was low. Accordingly, in the present study, grazing pressure was kept low, and ensured minimum ≈1 Mg ha^{-1} of standing biomass at all times and even spelled (no grazing) in later years (Thornton and Shrestha, 2020). For example, only 0.046 Mg N ha^{-1} or 1.5 kg N ha^{-1} y^{-1} was removed in animal products (beef) from this site over a 31 year period of grazing. This was compensated for by N addition via rainfall, estimated to be 2–17 kg ha^{-1} y^{-1} of N in this region (Hunter and Cowie, 1989; Packett, 2017). McSherry and Ritchie (2013) and Abdalla et al. (2018) found minimal effects on SOC stocks under low grazing intensity, which is similar to that observed by Pringle et al. (2014) for a Vertosol (Vertisol) under pasture grazed at low stocking rate over 26 y in a semi-arid region.

Although the total SOC stocks remained essentially similar from 1.75 to 33 y, despite the changes in C_3-forest to C_4-pasture, we estimated the changes in C_3-SOC and C_4-SOC stocks (Fig. 1b). After 33 y, the C_3-SOC

Table 2
Effect of land use change from forest to cropping on soil organic C (SOC) stocks, rate of SOC loss and turnover time at different depths in a Vertisol over 28 y prior to the introduction of a legume ley pasture.

Soil depth (m)	Initial SOC (Mg ha^{-1})	Final SOC[A] (Mg ha^{-1})	Rate of loss (1/year)	Turnover time (t)	R^2
0–0.1	27.9 ± 1.0	13.9 ± 0.6	0.119 ± 0.023	8.4 ± 1.6	0.98 **
0–0.2	40.7 ± 1.0	23.2 ± 0.6	0.125 ± 0.013	8.0 ± 0.8	0.99 **
0–0.3	53.5 ± 2.1	32.9 ± 1.0	0.141 ± 0.021	7.1 ± 1.1	0.98 **

** Significant at $P < 0.01$.
[A] SOC stock at steady state (final or equilibrium).

Table 3
Effect of land use change from forest to cropping on soil organic C (SOC) fractions, particulate organic C (POC), mineral-associated organic C (MAOC) and resistant organic C (ROC), and their rates of loss and turnover times at 0-0.1 m depth in a Vertisol over 28 y prior to the introduction of a legume ley pasture.

Soil depth (m)	Initial SOC (Mg ha^{-1})	Final SOC[C](Mg ha^{-1})	Rate of loss (1/ year)	Turnover time (year)	R^2
Particulate organic C (POC)					
0–0.1	6.9 ± 0.2	2.0 ± 0.2	0.175 ± 0.021	5.7 ± 0.7	0.99 **
0–0.2[A]	8.2 ± 0.6	na	0.186 ± 0.047	na	0.69*
0–0.3[A]	13.1 ± 0.9	na	0.236 ± 0.066	na	0.65*
Mineral-associated organic C (MAOC)					
0–0.1[B]	13.2 ± 1.7	7.7 ± 1.2	0.091 ± 0.047	11.0 ± 5.7	0.89 **
0–0.2	20.3 ± 1.3	12.2 ± 0.9	0.116 ± 0.037	8.6 ± 2.8	0.94 **
0–0.3	22.4 ± 1.3	16.5 ± 0.9	0.330 ± 0.098	4.4 ± 1.8	0.62*
Resistant organic carbon (ROC)					
0–0.1	7.8 ± 0.4	3.8 ± 0.3	0.089 ± 0.016	11.2 ± 2.0	0.99 **
0–0.2	10.9 ± 0.7	6.7 ± 0.5	0.119 ± 0.037	8.4 ± 2.6	0.94 **
0–0.3	14.3 ± 0.8	9.6 ± 0.5	0.195 ± 0.072	5.1 ± 1.9	0.86 **

[A] Linear rate of loss of particulate organic C (POC) at 0–0.2 m and 0–0.3 m depths.
[B] Equation for the linear rate of loss of mineral associated organic C (MAOC) at 0–0.1 m depth is: MAOC (Mg ha^{-1}) = 12.4 ± 0.5–0.193 ± 0.035 y; R^2 = 0.82**.
[C] SOC stock at steady state (final or equilibrium).
** Significant at $P < 0.01$; *Significant at $P < 0.05$.

stocks decreased by about 15 Mg C ha^{-1} whereas the C_4-SOC stocks increased by similar amounts, thus comprising 64 % of total SOC at 0–0.1 m depth (Fig. 1b). At steady state, the rate of C addition required ($A = C_e \times k$) was, according to C_3-SOC, 0.9 Mg C ha^{-1} y^{-1} (9 Mg C ha^{-1} × 0.105 y^{-1}), and according to C_4-SOC, 0.88 Mg C ha^{-1} y^{-1} (19.1 Mg C ha^{-1} × 0.046 y^{-1}) (Fig. 1b), that is, 0.9 Mg C ha^{-1} y^{-1} at 0–0.1 m depth. For a cropping site on a fine-textured Vertosol (Vertisol, 50 % clay), Dalal and Mayer (1986a) estimated that 0.7 Mg C ha^{-1} y^{-1} at 0–0.1 m depth was required to maintain SOC at steady state, a lower estimate possibly due to its higher clay content (>50 % clay) than at the present site (36 % clay).

Root biomass is the main source of C input to SOC (Rasse et al., 2005) via soil microbial C (Kramer et al., 2010). From only one period of monitoring at this site, Dalal et al. (2013) estimated 4 Mg ha^{-1} of root biomass C to maintain SOC at 0–0.4 m depth at steady state. Robertson et al. (1993) measured the root biomass C turnover of about 5 Mg ha^{-1} over a 2-year period at a Vertisol site or 2.5 Mg ha^{-1} y^{-1} at 0–28 cm, mostly at 0–0.1 m depth. The total aboveground pasture biomass production was 238 Mg ha^{-1} at this site (see earlier). Belowground root biomass production was estimated to be 2.5 times of this amount and contained 40 % C (Robertson et al., 1993), or 7.4 Mg ha^{-1} of root C biomass over a 31 y pasture period, thereby exceeding the earlier estimates of Dalal et al. (2013). These values are also outside the range (1.9–4.9 Mg C ha^{-1} at 0–0.3 m depth) estimated by Allen et al. (2016) for pasture in the northern Brigalow belt bioregion, Queensland, Australia. It is likely that substantial amount of pasture root biomass C inputs may have occurred below 0.3 m depth in this uniform-textured clay Vertisol. Dalal et al. (2005a) found that C_4-buffel grass pasture after 20 years contributed 19.3 % of total SOC at 0.3–0.6 m depth and 19.2 % at 0.6–1.0 m depth in a Red Kandosol (Profondic Lixisol), which originally supported C_3-mulga (*Acacia aneura* L.) vegetation.

Not only did we observe that total SOC stocks remained relatively constant upon conversion to pasture, but we also observed that the contributions of the various SOC fractions did not change significantly ($P > 0.05$) with time (Fig. 1c), nor did the C:N ratio (Fig. S3), thus generally retaining SOM quality. This is in contrast to the prediction of Baldock et al. (2013a). In Australian soils, the ROC fraction is considered to be the most biologically stable form of organic C and is dominated by char-type material derived from incomplete combustion of plant residues (Hoyle et al., 2011). These findings are somewhat consistent with the findings of Wilson and Lonergan (2013), with these authors examining soils in New South Wales (Australia) and finding that ROC stocks were similar across improved pastures, native pastures, and lightly wooded pastures (although there was no comparison with native vegetation). However, few studies have examined long-term impacts of land use change to pasture on these SOC fractions.

Our observation that the contributions of the various SOC fractions did not change significantly ($P > 0.05$) with time (Fig. 1c) is likely because the SOC fractions we measured are not sensitive enough to detect the qualitative changes in SOC that occur on conversion of native forest to pasture. For example, Dalal et al. (2013) reported that although the total SOC stocks in soil under the native forest and pasture were similar, the >250 μm aggregate fraction contained only 12 % of the SOC under pasture compared to 43 % under native forest, resulting in a loss of 70 % of the SOC from >250 μm aggregate fraction. The <53 μm fraction, however, contained 70 % of the total SOC under pasture but only 41 % under native forest at 0–0.1 m depth. As suggested by Dungait et al. (2012) and Liang et al. (2017) that SOM turnover is governed by microbial-enzyme accessibility to the substrate rather than recalcitrance, the nature of the SOC fractions (POC, MAOC and ROC) measured in this study may not be sensitive enough to detect changes in the nature of SOC due to land use change from native forest to pasture.

4.2. Soil organic C and soil total N dynamics following land use change from forest to cropping

Land use change from native forest to arable cropping resulted in a pronounced decrease in both SOC (48 %) and STN (42 %) within 26 y (1984–2010) at 0–0.1 m depth in this Vertisol (Vertosol) before the ley pasture phase was initiated (Fig. 3b). These findings confirm earlier results from this region and elsewhere that long-term cropping often decreases SOC and STN stocks by 30–60 % (Zach et al., 2006; Luo et al., 2010; Don et al., 2011; Beniston et al., 2014; Kopittke et al., 2017). For example, Dalal and Mayer (1986a) and Dalal and Mayer (1986b) found decreases in both SOC (65 %) and STN (57 %) at 0–0.1 m depth in a Vertosol (Vertisol) that supported similar native forest (brigalow) prior to cropping for 45 y. Similarly, Gregory et al. (2016) measured losses of SOC (65 %) and STN (61 %) at 0–0.15 m depth after 59 y of land use change from grassland to arable cropping in the UK. In the present study, at 0–0.3 m depth, over a 28 y period prior to the introduction of a legume ley pasture, the decrease in total SOC was 20.6 Mg ha^{-1} (39 %)

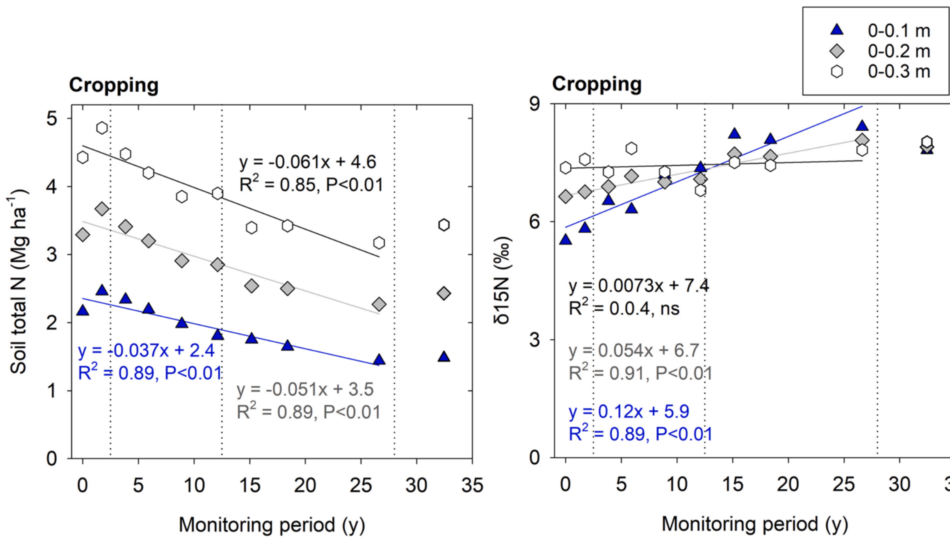

Fig. 4. Changes in soil total N (STN) stocks and δ^{15}N at 0-0.1 m, 0-0.2 m, and 0-0.3 m depths, following land use change from native forest to cropping for 33 y (1981-2014). The first crop (sorghum) was planted 2.5 y after pulling and clearing commenced. Thereafter, Phase 1 of cropping was a wheat monoculture. Phase 2 of cropping started 12 y after pulling and clearing commenced, with 10 sorghum crops, four wheat, one barley, and one chickpea. After 28 y, a legume ley pasture was sown (excluded from regression analysis).

Table 4
Effect of land use change from forest to cropping on soil total N (STN) stocks, rate of STN loss and turnover time at different depths in a Vertisol over 28 y prior to the introduction of a legume ley pasture.

Soil depth (m)	Initial STN (Mg ha^{-1})	STN at 28 y (Mg ha^{-1})[A]	Change in STN (Mg ha^{-1} y^{-1})	R^2
0–0.1	2.35 ± 0.07	1.43 ± 0.33	−0.037 ± 0.005	0.89 **
0–0.2	3.49 ± 0.09	2.26 ± 0.42	−0.051 ± 0.007	0.89 **
0–0.3	4.60 ± 0.12	3.17 ± 0.52	−0.061 ± 0.009	0.87 **

** Significant at P < 0.01; *Significant at P < 0.05.
[A] Measured STN.

and the decrease in STN was 1.7 Mg ha^{-1} (37 %) (Figs. 3 and 4 and Table 2); these values are similar to those reported by Don et al. (2011) for tropical soils at this depth.

Apparently, the amount of plant C and N added to the soil during cropping was insufficient to maintain SOC at this site. Thornton and Shrestha (2020) measured 49 Mg ha^{-1} grain yield over the monitoring period. Assuming harvest index of 0.4, a root biomass: aboveground biomass ratio of 0.2 and 40 % C concentration of biomass (Robertson et al., 1993), gives 9.8 Mg ha^{-1} of root biomass C or 0.3 Mg ha^{-1} y^{-1} root biomass C. However, this was <20 % of the C addition required (1.7 Mg C ha^{-1} y^{-1}) to maintain SOC at steady state at this site at 0–0.1 m and ca. 10 % of the 2.9 Mg C ha^{-1} y^{-1} (23.2 Mg ha^{-1} × 0.125 y^{-1}) required to maintain SOC at steady state at 0–0.2 m depth (Table 2). The latter value is similar to that required to maintain the SOC stocks under pasture in this study. Robertson et al. (1994) measured 4.5 Mg ha^{-1} y^{-1} of C turnover through microbial biomass under sorghum grown on a Vertosol (Vertisol). However, larger amounts of root biomass C may be required to maintain SOC stocks at steady state at 0–0.3 m depth (4.6 Mg C ha^{-1} y^{-1}). Continuing cropping at this site, therefore, will likely lead to continuing decreases of SOC stocks, as supported by the modelling of Huth et al. (2010). The legume ley pasture, butterfly pea sown at this site 4 y prior to the end of the monitoring period may arrest further decrease in SOC given that N_2-fixed legume N may also slow down the rate of SOC loss and turnover (Ramirez et al., 2010; Finn et al., 2016) in addition to C inputs and no N export in grain harvest. Alternatively, the SOC stocks could be increased by increasing crop root biomass and N fertilisation or legume N inputs in the cropped soils (Chenu et al., 2019) although N_2O emissions may also increase over time (Dalal et al., 2003), thus partly reducing the impact of SOC increase on climate change mitigation (Huth et al., 2010; Dalal et al., 2013).

Not only did we observe a marked decrease in SOC, but STN also decreased by 0.037 ± 0.005 Mg ha^{-1} y^{-1} at 0–0.1 m depth during the first 28 y prior to the introduction of a ley pasture (Fig. 4a). This decrease in the STN stock was similar to the N removed in the grain harvest (0.958 Mg ha^{-1}), at an average rate of 0.030 Mg ha^{-1} y^{-1} (Thornton and Shrestha, 2020). However, the rate of decrease in STN at 0–0.3 m depth during the first 28 y was 0.061 ± 0.007 Mg ha^{-1} y^{-1} (Table 4). Most of this loss in the STN may have been due to runoff and soil erosion, and N_2O emissions in the earlier years (Hunter and Cowie, 1989; Cowie et al., 2007; Dalal et al., 2013; Elledge and Thornton, 2017), and deep drainage during the fallow periods at least until 2010 before the introduction of butterfly pea as a ley pasture. Indeed, at the cropping site, almost 20 mm y^{-1} of deep drainage occurred versus ≤ 1 mm y^{-1} from the adjacent native forest site (Silburn et al., 2009). It is likely that introduction of ley pasture to the cropping site may reduce this loss of SOC that has resulted previously from the low C inputs and the disturbance of soil from tillage operations, as well as reducing the STN losses that were caused by grain N harvest as well as deep drainage. Indeed, the observation that there were apparently no further decreases in both the SOC and STN stocks observed during the last 4 y of monitoring (Fig. 4) supports this hypothesis.

There were a total of 26 crops harvested between 1984 and 2010 at this site (Thornton and Shrestha, 2020); of these, 11 crops were C_4-crops. The total grain yield of C_4-crops was 20.8 Mg ha^{-1}. Using harvest index of 0.4, a root biomass: aboveground biomass ratio of 0.2 and 40 % C concentration (Robertson et al., 1993), 10.4 Mg ha^{-1} of root biomass or 4.2 Mg ha^{-1} of C_4–C was added over a 28 y period prior to the introduction of the ley pasture. The measured C_4-SOC increase of 0.11 Mg ha^{-1} y^{-1} or 3.1 Mg ha^{-1} during the monitoring period at 0–0.1 m depth, alone accounts for 74 % of the C_4-SOC (Fig. 3b); thus, most of the root biomass C was retained as the C_4-SOC at the site.

We also examined changes in the three SOC fractions upon conversion to cropping, hypothesising that ROC stocks would remain relatively constant over time despite decreases in total SOC stocks. However, in contrast to that observed for pasture, the decreases in SOC upon commencement of cropping at this site also occurred in all three SOC fractions, POC, MAOC and ROC (Fig. 3b and Fig. S4). Although the POC fraction decreased at a faster rate than the MAOC and ROC fractions, both MAOC and ROC decreased markedly over time (Fig. 3c and Table 3). The SOC modelling at this site using RothC (Skjemstad et al., 2004) and APSIM (Luo et al., 2014) and including ROC fraction as the inert C in the model, was presumed to be successful. The assumption that the ROC is an inert C fraction in these models is inconsistent with our findings. This was also confirmed by Page et al. (2013) for another Vertisol (Vertosol, 65 % clay) used for cereal cropping, where the authors found that all the SOC fractions decreased over a 27 y period. In a similar manner, Briedis et al. (2020) also found that the ROC fraction was altered by land use change from native vegetation to cropping. Obviously, SOM turnover is largely governed by microbial-enzyme accessibility (Dungait et al., 2012; Liang et al., 2017) rather than recalcitrance such as the ROC fraction. Dalal and Mayer (1986a) also found that the rate of decrease of SOC under cropping was negatively related to aggregation (index) and positively related to urease activity, thereby suggesting the importance of microbial-enzyme accessibility for SOM turnover. There is likely to be much greater accessibility to SOM by microbes and enzymes under cropping due to aggregate disruption than pasture, which is supported by the findings in this study. Introduction of the legume ley since 2010 may increase (or at least maintain) SOC at this site by reducing soil disturbance and N removal in grain as well as potentially increasing legume-N inputs, thus slowing SOM turnover.

5. Conclusions

Maintenance of SOM is essential for soil fertility, productivity, environmental sustainability and climate change mitigation. We measured SOC and STN following land use change from native forest to either pasture or cropping over a 33 y period. We found that land use change from the native forest to pasture decreased the SOC stocks by 12.2 % and STN stocks by 22.6 % during the land development to pasture establishment phase (<1.75 y), although SOC and STN stocks at the study sites remained relatively constant thereafter due mainly to pasture root biomass C inputs and relatively low N removal in animal product. Also, stocks of the various SOC fractions, POC, MAOC and ROC, did not change with time under pasture. In contrast to these modest changes for pasture, land use change to cropping led to large decreases in the SOC (48 %) and STN (42 %) stocks at 0−0.1 m depth within 28 y (1981–2010), due mainly to insufficient C inputs and N export in grain harvest exceeding the level required to maintain SOM at steady state. Moreover, stocks of all the SOC fractions (including ROC) decreased with time, indicating that the ROC fraction was not a recalcitrant fraction under cropping. Our results indicate that the biomass C inputs by the crops, mainly as root biomass C, were not sufficient to reverse or slow down the rate of decrease of SOC and STN in this soil. However, importantly, this loss of SOC (and its fractions) and STN caused by conversion to cropping was arrested by the introduction of the ley pasture phase during the last 4 y of monitoring. These findings regarding the changes in SOC and STN stocks for a low input system in semi-arid to subtropical Australia are important, especially for the ROC fraction which is generally assumed to be largely inert and not susceptible to changes as a result of land use change followed by long-term cropping.

Declaration of Competing Interest

The authors declare no conflict of interest.

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Appendix A. Supplementary data

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