Jacqueline R. England, Keryn I. Paul, Shaun C. Cunningham, Dinesh B. Madhavan, Thomas G. Baker, Zoe Read, Brian R. Wilson, Timothy R. Cavagnaro, Tom Lewis, Michael P. Perring, Tim Herrmann, Philip J. Polglase

Previous land use and climate influence differences in soil organic carbon following reforestation of agricultural land with mixed-species plantings

Agriculture, Ecosystems and Environment, 2016; 227:61-72

Final publication at http://dx.doi.org/10.1016/j.agee.2016.04.026

Previous land use and climate influence differences in soil organic carbon following reforestation of agricultural land with mixed-species plantings

Jacqueline R England[1], Keryn I Paul[2], Shaun C Cunningham[3, 4], Dinesh B Madhavan[5], Thomas G Baker[5], Zoe Read[6], Brian R Wilson[7], Timothy R Cavagnaro[8], Tom Lewis[9], Michael P Perring[10, 11], Tim Herrmann[12], Philip J Polglase[2,6]

[1] CSIRO Agriculture and CSIRO Land and Water, Private Bag 10, Clayton South, VIC 3169, Australia;

[2] CSIRO Agriculture and CSIRO Land and Water, GPO Box 1700, ACT 2601, Australia;

[3] School of Life and Environmental Sciences, Deakin University, Burwood, VIC 3125, Australia;

[4] Institute for Applied Ecology, University of Canberra, Bruce, ACT 2617, Australia

[5] School of Ecosystem and Forest Sciences, The University of Melbourne, Richmond, VIC 3121, Australia;

[6] Fenner School of Environment and Society, Australian National University, Canberra, ACT 0200, Australia;

[7] School of Environmental and Rural Sciences, University of New England, Armidale, NSW 2351, Australia;

[8] School of Agriculture, Food and Wine, University of Adelaide, Waite Campus, PMB 1 Glen Osmond, SA 5064, Australia;

[9] Agri-Science Queensland, Department of Agriculture, Fisheries and Forestry, QLD 4556, Australia;

[10] School of Plant Biology, The University of Western Australia, Crawley, WA 6009, Australia;

[11] Forest and Nature Lab, Ghent University, BE-9090, Gontrode-Melle, Belgium;

[12] Department of Environment, Water and Natural Resources, South Australia, Australia.

Corresponding author: jacqui.england@csiro.au

Abstract

Reforestation of agricultural land with mixed-species environmental plantings (native trees and shrubs) can contribute to mitigation of climate change through sequestration of carbon. Although soil carbon sequestration following reforestation has been investigated at site- and regional-scales, there are few studies across regions where the impact of a broad range of site conditions and management practices can be assessed. We collated new and existing data on soil organic carbon (SOC, 0-30 cm depth, N = 117 sites) and litter (N = 106 sites) under mixed-species plantings and an agricultural pair or baseline across southern and eastern Australia. Sites covered a range of previous land uses, initial SOC stocks, climatic conditions and management types. Differences in total SOC stocks following reforestation were significant at 52% of sites, with a mean rate of increase of 0.57±0.06 Mg C ha^{-1} y^{-1}. Increases were largely in the particulate fraction, which increased significantly at 46% of sites compared with increases at 27% of sites for the humus fraction. Although relative increase was highest in the particulate fraction, the humus fraction was the largest proportion of total SOC and so absolute differences in both fractions were similar. Accumulation rates of carbon in litter were 0.39±0.02 Mg C ha^{-1} y^{-1}, increasing the total (soil + litter) annual rate of carbon sequestration by 68%. Previously-cropped sites accumulated more SOC than previously-grazed sites. The explained variance differed widely among empirical models of differences in SOC stocks following reforestation according to SOC fraction and depth for previously-grazed (R^2 = 0.18-0.51) and previously-cropped (R^2 = 0.14-0.60) sites. For previously-grazed sites, differences in SOC following reforestation were negatively related to total SOC in the pasture. By comparison, for previously-cropped sites, differences in SOC were positively related to mean annual rainfall. This improved broad-scale understanding of the magnitude and predictors of changes in stocks of soil and litter C following reforestation is valuable for the development of policy on carbon markets and the establishment of future mixed-species environmental plantings.

Keywords: labile organic carbon; resistant organic carbon; humus organic carbon; plant litter; carbon sequestration; environmental plantings; revegetation

1. Introduction

There is increasing interest in reforestation of agricultural lands to sequester carbon in woody biomass and potentially mitigate greenhouse gas emissions (e.g. Canadell and Raupach, 2008; Cunningham *et al.*, 2015a). Reforestation can increase terrestrial carbon through humification and storage in soil organic carbon (SOC; Lal, 2005). While reforestation of agricultural lands significantly increases carbon sequestration in biomass compared with crop or pasture (e.g. Paul *et al.*, 2008; Cunningham *et al.*, 2015b), changes in SOC following reforestation are highly variable and uncertain, with increases, negligible change and decreases reported (e.g. Specht and West, 2003; Lima *et al.*, 2006; Harper *et al.*, 2012). Some variation may be explained by time since reforestation, as generally there are initial decreases in SOC stocks in the first five years after reforestation, followed by recovery to pre-establishment levels (approx. 10-30 years), and then a gradual increase (e.g. Paul *et al*., 2002). However, most of the variability in SOC stocks following reforestation reflects differences in sequestration rates among climates, soil types, tree species and previous land uses (Guo and Gifford, 2002; Paul *et al.*, 2002; Laganière *et al.*, 2010).

Previous land use can be an important determinant of sequestration of SOC following reforestation, with increases in stocks on ex-cropland and predominantly losses on ex-pasture (Guo and Gifford, 2002; Laganière *et al.*, 2010). Climate can have a strong influence, with increases in tropical and sub-tropical regions compared with small decreases in temperate and Mediterranean-type regions (e.g. Paul *et al.*, 2002). Soils with high clay content generally having a larger capacity to accumulate SOC than those with lower clay content (Laganière *et al*., 2010; Paul *et al*., 2002). Further, the tree species planted can affect carbon sequestration, with increases in SOC stocks under some nitrogen-fixing acacia trees (Kasel *et al.*, 2011; Forrester *et al.*, 2013), although these effects may be species-specific (Hoogmoed *et al.*, 2014), whereas decreases in SOC stocks have been found under pines (Parfitt *et al.*, 1997; Turner and Lambert, 2000).

Accumulation of plant litter is an additional store of carbon in forests until a steady state between litterfall and decomposition is reached (Paul *et al.*, 2003). Rates of litter accumulation in

native and plantation forests differ widely among forest types and species (e.g. Spain, 1984; Adams and Attiwill, 1991; Fernandez-Nunez *et al.*, 2010), predominantly reflecting differences in litter quality and climate (Prescott, 2010). Pine plantations can accumulate particularly thick and recalcitrant litter layers (e.g. Paul *et al.*, 2003; Paul and Polglase, 2004) compared with those under other plantation species (e.g. Turner, 1986; Harper *et al.*, 2012). Comparable studies under mixed-species plantings are limited, but suggest that thick layers of up to approximately15 t dry matter (DM) ha^{-1} can accumulate within two decades when eucalypts are planted (Cunningham *et al.*, 2012).

SOC exists as a diverse mix of organic materials with different susceptibilities to biological decomposition (Baldock *et al.*, 2013a). Reforestation may change the molecular form of SOC and, consequently, increase the stability of the stock (Cunningham et al., 2015b). For a given organic carbon content, the provision of energy to soil organisms should increase with increasing proportion of plant litter-like components and decrease with increasing proportion of biologically-recalcitrant charcoal or char-like components (Baldock *et al.*, 2013a). Reforestation may increase inputs of more resistant SOC to soil but generally there is little increase in resistant humic material within three decades, although earlier increases have been observed (e.g. Del Galdo *et al.*, 2003; Cunningham *et al.*, 2015b). Understanding the form of SOC sequestered after reforestation (i.e. its stability) is important in predicting the longer-term rates of sequestration and resilience of carbon stocks to future change (e.g. with climate change), and to calibration and verification of process-based models of turnover and accumulation of SOC.

Establishment of mixed-species environmental plantings (i.e. plantings of native tree and shrub species established for environmental benefits with no intention to harvest) on agricultural land can be an economically-viable option in lower rainfall (< 1000 mm y^{-1}) regions (Crossman *et al.*, 2011; Polglase *et al.*, 2013). Indeed, environmental plantings are increasingly being established to sequester carbon because of their co-benefits to the environment and biodiversity (e.g. Mitchell *et al.*, 2012). Consequently, measurement and modelling of biomass carbon in

environmental plantings across a broad range of climatic and management conditions has been the focus of recent work (e.g. Paul *et al.*, 2015). In contrast, there are limited measurements of changes in soil and litter carbon under such plantings (e.g. Cunningham *et al*., 2015b), and little is known about their potential to sequester carbon in litter and soil compared with production forests (Cunningham *et al*. 2015a). Global meta-analyses of soil carbon sequestration following reforestation (Silver *et al.*, 2000; Guo and Gifford, 2002; Paul *et al.*, 2002; Laganière *et al.*, 2010) include few studies of mixed-species plantings, and even meta-analyses of biomass accumulation in mixed-species plantings have been dominated by plantings with only two species (Piotto, 2008; Hulvey *et al.*, 2013). Further, environmental plantings are highly variable, being established across a much broader range of climates, previous land uses and landscape positions than a given commercial plantation type (Paul *et al*., 2015).

Here, we assessed potential predictors of soil carbon sequestration under environmental plantings, which will inform their future establishment, calibration of carbon accounting models and development of policy on carbon markets. A national dataset of 117 Australian sites was collated and analysed, which represented much of the temperate and Mediterranean-type climates across the continent. Three key research questions were addressed in relation to changes in carbon sequestration following reforestation with environmental plantings:

1) Are there significant differences in stocks of total SOC and its fractions (particulate, humus, resistant)?

2) Are estimates of carbon sequestration significantly increased when stocks of litter are included?

3) What are the key site conditions and management practices that determine the magnitude of differences in SOC and litter stocks?

2. Methods

2.1 Study sites

New and existing data were collated from 117 mixed-species environmental plantings (subsequently termed 'environmental plantings') established on agricultural land. Plantings were across southern and eastern Australia (latitude -30.9 to -38.7 °S, longitude 117.4 to 150.3 °E), and covered the range of rainfall zones where planting occurs (380 to 1147 mm y^{-1}; Table 1, Fig. 1). Thirty-six new sites were measured to improve the representativeness of plantings with respect to age, previous land use, productivity (aboveground biomass increment) and soil texture (Table 1). Environmental plantings are often established along stream banks to reduce erosion or in areas with shallow water tables to mitigate dryland salinity by minimising recharge (George *et al*., 1999). Thus sites were selected to include both dryland plantings across a range of landscape positions (N = 97) and riparian plantings (N = 20).

Previous land use at the sites included grazing, cropping, and rotational cropping and grazing (Table 1). For analysis, we combined cropping with rotational cropping and grazing as 'cropping'. There were fewer previously-cropped sites (N = 32) than previously-grazed sites (N = 85) reflecting current reforestation activity in Australia. Because environmental plantings are a relatively new land use, mean age of the plantings was 14 y, with 95% of plantings aged between 1 and 28 y. Biomass productivity varied greatly among sites (0.2 to 31.7 Mg DM ha^{-1} y^{-1}, calculated using methods of Paul *et al*., 2015) due to differences in site characteristics (e.g. soil type, rainfall), planting structure (e.g. stand density, species-mix, planting configuration) and age.

2.2 Soil and litter sampling at existing sites

Existing data on total organic carbon (TOC) in soil under environmental plantings were collated from 81 sites in temperate Australia. These data were collected between 2009 and 2012 using varying protocols including plot- and transect-based sampling. Studies limited to locations directly under trees were excluded because they are not representative of soil from across all

planting strata (i.e. under trees, between trees along rows, inter-rows). Environmental plantings were paired with the adjacent agricultural land use at 64 sites, whereas there had been a prior baseline sampling (i.e. measurement before and after reforestation) at the remaining 17 sites.

Soil was sampled to 30 cm depth from either four (0-5, 5-10, 10-20, 20-30 cm) or two depths (0-10 and 10-30 cm, or 0-5 and 5-30 cm depths) using a corer (internal diam. 42-73 mm). At each site, 5-18 replicates (each comprising 1-6 cores) from 1-8 plots/transects were collected. Cores were retained individually or bulked into 1-2 composite samples per transect, providing 5-18 samples per depth for the planting and 1-10 samples per depth for the adjacent agricultural land for analysis.

Existing data on plant litter (≤ 25 mm diam.) were collated from 53 of these sites (Table 1). At each site, 3-20 replicates from 1-36 quadrats (0.06-0.25 m^2) per plot/transect were sampled, with quadrats processed either separately or bulked.

2.3 2.2 Soil and litter sampling at new sites

New data on SOC and plant litter under environmental plantings and agricultural pairs were collected using standard protocols from an additional 36 sites and repeat sampling at the 17 existing baseline sites (N = 53 sites) in 2013. Sites were selected to fit the criteria that plantings were: i) at least 5 years old, ii) established on previous agricultural land, and iii) paired adequately with the adjacent agricultural land with respect to soil type, landscape position and land-use history prior to planting.

The standard protocol (at 36 new 'standard' sites) involved sampling soil and litter from a representative 0.4 ha area within the planting, and within the adjacent agricultural field. The 0.4 ha plot was divided into 40 sampling units of 10 m × 10 m, with one sampling position randomly located per unit. For belts (10-40 m wide), sampling units were across the entire belt and for blocks (> 40 m wide) a buffer of at least 5 m was left on each edge to minimise edge effects. Both plots in the planting and agricultural pair were aligned along the same contour to minimise

differences in soil type and management history among the corresponding 40 sampling units in the two land uses. Cores were sampled randomly within each unit to ensure that variation at sub-plot scales due to past management practices (e.g. ripping and sowing lines, VandenBygaart, 2006) was captured in the estimate for the whole plot. Three additional sites were sampled more intensively ('intensive' sites) as part of a study investigating the adequacy of sampling intensity (Cunningham *et al.*, in review).

Soil was sampled using a corer (internal diam. 40-72 mm) to 30 cm depth with a minimum of two depth intervals (0-10 and 10-30 cm), but with four depths (0-5, 5-10, 10-20, 20-30 cm) and three depths (0-5, 5-10 and 10-30 cm) at some sites. Plant litter was sampled from a quadrat (0.1-0.25 m^2) above each soil sampling location. At the standard sites, for each of the soil depths and the litter layer, five randomly-selected cores/quadrats were bulked into one composite sample. This provided eight composite samples for each soil depth for both the planting and for the adjacent agricultural land. At the intensive sites, for each of the two soil depths and the litter layer, 120 cores/quadrats in the planting and 56 in the adjacent agricultural land were analysed individually for TOC/litter mass. Soil samples from the intensive sites were bulked randomly into eight composite samples per depth (from 16 samples in the planting and seven in the adjacent agricultural land) for analysis of the particulate (POC), humus (HOC) and resistant (ROC) organic carbon fractions.

2.4 Sample processing

Soil samples were air-dried and sieved to ≤ 2 mm. High clay content soils were crushed (predominantly using a jaw-crusher) prior to sieving to ensure that aggregates were not retained in the > 2 mm fraction. Representative subsamples for analysis were obtained by careful mixing either using a riffle box (Civilab, 12 × 13 mm slotted box) or by quartering. Air-dried litter samples were separated into two components, fine (< 2 mm) and coarse (> 2-25 mm), by passing through a 2 mm sieve. For 47 of the 53 existing sites with litter data, only a total litter mass was

recorded. Subsamples were oven-dried at 60-70 ^{o}C to constant mass. Total litter stocks were calculated in Mg DM ha^{-1}.

2.5 Analysis of soil carbon content

TOC concentration was determined on a subsample of finely ground < 2-mm air-dried soil using a CNS-2000 or CNS TruMac analyser (LECO Corporation, St. Joseph, MI, USA). For 35 of the existing sites (Cunningham *et al.*, 2015b), C concentrations of each sample were determined from a subsample of 4–5 mg using catalytic combustion and thermal conductivity (vario MICRO cube, Elementar Analysensysteme GmbH, Hanau, Germany). At a further 20 existing sites, a wet oxidation method (Heanes, 1984) was used to determine TOC concentration. Where appropriate, samples were fizz-tested for the presence of carbonates with a few drops of 1 M HCl placed directly on the sample before analysis. Positive fizz samples were re-analysed with the elemental analyser after pre-treatment with H_2SO_3 as a 5–6 wt% SO_2 solution to remove inorganic carbon.

The POC, HOC and ROC fractions for all soil samples were predicted using mid-infrared spectroscopy (MIRS) following the methods of Baldock *et al*. (2013a). The predictions were made using new calibrations (Madhavan *et al.*, 2016) incorporating soils from environmental plantings (largely this study) and agricultural soils (Baldock *et al.*, 2013a).

The moisture content in the < 2 mm soil samples after air drying was quantified gravimetrically by oven drying a 30-40 g sample at 105 °C for 16-24 h. To estimate SOC contents from soil carbon concentrations, volumetric soil mass (g cm^{-3}) of the 0-10, 10-30 and 0-30 cm soil depths was calculated from the mass of the fine (< 2 mm) fraction and volume of the bulked samples corrected for oven-dry (105 °C) moisture content. TOC, POC, HOC and ROC stocks were calculated in Mg ha^{-1} for these fixed depths. Stocks of TOC were corrected for equivalent soil mass following the equation given in Wendt and Hauser(2013). To do this, the soil mass to 30 cm depth was calculated for all replicates and the 10th percentile of the masses

determined. The proportion of the soil mass required from the deepest depth interval (either 20–30 cm or 10-30 cm) to achieve the target mass was applied to determine the mass of carbon included from this depth increment.

2.6 Analysis of litter carbon content

A subset of 298 litter samples were selected for analysis of C content, including fine and coarse fractions, from across 29 standard sites. Total C concentration was determined on a subsample of finely-ground litter using a CNS-2000 or CNS TruMac analyser (LECO Corporation, St. Joseph, MI, USA). Across the broader dataset (*N* = 106 sites), C stock of each litter component was calculated in Mg C ha^{-1} either by multiplying the litter mass with the measured C concentrations for the site (*N* = 29 sites), or for sites where litter C was not analysed, using mean C concentrations (*N* = 88 sites).

2.7 Dataset collation

2.7.1 Predictor variables

To assess potential predictors of differences in SOC following reforestation, we collated a range of site variables that were potential predictors. Collection of stand, management, climate and soil variables are described below, with values of final candidate predictor variables used in generalized linear models (see Section 2.8.2) given in Table 2.

Long-term (45-year; 1968-2012) monthly climate data for each of the sites were extracted from the SILO Data Drill (Jeffrey *et al.*, 2001; https://www.longpaddock.qld.gov.au/silo/). We derived both long-term climate variables and climate variables since planting (Table 2). Surrogate data for climate, including elevation, latitude and longitude, were included as predictor variables.

Soil types and textures for each site were obtained using GIS layers of the Australian Soil Classification system of the Atlas of Australian Soils (ABARES, 2004) and textural classes (McKenzie *et al.*, 2000). At the broadest classification, the majority of the sampled soils (68%)

were Sodosols, so this was included as a predictor (Table 2). Soil texture was a categorical variable with values of 1 (sand), 2 (sandy loam), 3 (loam), 4 (clay loam) or 5 (clay). Soil clay content was obtained from spatial layers underpinning the national carbon accounting model FullCAM (DIICCSRTE, 2014).

Land managers provided information about the age of the planting at the time of measurement and the previous land use. Plantings were either planted as tubestock or direct seeded, and we included a predictor for tubestock planting. Mean planting width was measured on site or estimated from imagery in Google^{TM} Earth.

Measurement plots (*N* = 1-51 per planting, area = 0.01-3.37 ha) or transects (*N* = 2-8 per planting, area = 0.001-0.010 ha) were established at each site to estimate biomass and stand structure. Plots/transects were surveyed to coincide with soil sampling (2013 and 2009-2012, respectively). Within each plot, stem diameters of all trees (at 130 cm height) and shrubs (at 10 cm height) were measured. Above-ground (AG) biomass for individual trees or shrubs was estimated by applying existing species-specific allometric equations developed from environmental plantings in temperate Australia (Paul *et al.*, 2013) and scaled to a live AG biomass (Mg DM ha^{-1}) for each planting (Paul *et al*., 2015). Stand density was calculated as the number of trees/shrubs per hectare. The proportion of trees, proportion of eucalypts (predominant trees in plantings), proportion of acacias (dominant understorey in most plantings) and the proportion of potential nitrogen-fixing trees (trees belonging to the genera *Acacia*, *Allocasuarina* and *Casuarina*) were calculated based on stand density (Table 2). Proportions of tree types were based on density as proportions based on live AG biomass were found to be poorer predictors (data not shown).

Many riparian plantings, and some plantings from lower landscape positions, had exceptionally high rates of accumulation of AG biomass compared with a recent national survey (Paul *et al*., 2015), suggesting access to additional water. Consequently, whether a planting was

riparian, was established in an upper landscape position or was assumed to have access to additional water were included as predictor variables (Table 2).

2.7.2 Adequacy of sampling intensity

SOC is often highly variable (coefficient of variation = 0.1–0.6, Wilson *et al.*, 1997; Paz-Gonzalez *et al.*, 2000), so adequate sampling of different soil depths, cores within plots and plots across a site, is required to provide accurate estimates (Cunningham *et al.*, 2012). Concurrent work on adequate sampling intensity at the intensively-measured sites (Cunningham *et al*., in review) found that a minimum of 30 cores were required to have a 95% probability of estimating SOC within 10% of the population mean. We assessed differences at all individual sites (N = 117) and then compared these differences with only sites with adequate sampling intensity (i.e. > 30 cores per site; N = 71 sites; see Section 2.8.1 below). However, only sites with adequate sampling intensity were included in analyses of potential predictors of SOC following reforestation (see Section 2.8.2 below). The dataset was further reduced by excluding one site with unexplained very high carbon stocks (ca 170 t C ha^{-1} to 30 cm depth), resulting in N = 70, 32 and 38 sites for analysis of all sites, previously-cropped and previously-grazed sites, respectively.

2.8 Data analysis

2.8.1Differences in SOC between land uses at individual sites

For each site, we used a *t*-test to assess whether TOC, POC, HOC or ROC stocks differed significantly between the paired mixed-species planting and agricultural land use by depth interval (0-10, 10-30, 0-30 cm). Differences between land uses were used to infer changes in SOC following reforestation, with the assumption that the agricultural land use was in steady state (i.e. any differences in SOC were attributable to the plantings). The number of samples available for statistical analysis depended on whether the site was: i) an existing site (N = 5-10 composite samples or individual cores), ii) a new standard site (N = 8 composite samples) or iii) a

new intensive site (N = 56-120 individual cores). The dominant component of the resistant fraction (ROC) measured here is charcoal (Baldock *et al.*, 2013b). Because no difference in ROC would be expected over the time since reforestation covered in this study (Skjemstad and Spouncer, 2003; Skjemstad *et al.*, 2004), ROC was not included in subsequent analyses of the key factors influencing differences in SOC.

2.8.2 Key factors influencing differences in SOC

We tested whether difference in SOC stock (calculated as SOC in the planting minus SOC in the agricultural pair/baseline) across sites was related to known predictors of change following reforestation (i.e. planting age, SOC stock in the agricultural pair/baseline, annual rainfall and previous land use) using linear regression. The exception to this was previous land use, which was categorical (either 'cropping' or 'pasture'), so *t*-tests were used. The only significant linear regression was between HOC and age, albeit a poor relationship (R^2 = 0.05, F = 4.93, P = 0.03, N = 70). Overall, difference in SOC differed significantly between previously-cropped and previously-grazed land in the surface soil (df = 68, t = 2.09, P = 0.04). Further, because both 'reference' SOC stock (i.e. TOC in the agricultural pair or baseline) and MAR differed significantly (degrees of freedom (df) = 68, t > 2.33, P < 0.05) between previously-cropped and previously-grazed land uses, in subsequent analyses, the dataset was separated into previously cropped (N = 32) and previously grazed (N = 38) sites.

Generalized linear modelling (GLM) with Bayesian model averaging (BMA) was performed using the ‘bic.glm’ function in the ‘BMA’ package of R (Raftery *et al.*, 2008) to determine what stand, management, climate and soil variables were strong predictors of difference in SOC (TOC, POC, HOC) stocks. BMA involves fitting multiple models and finding predictors with the highest posterior probabilities of inclusion in good models. Predictor variables with a probability of inclusion [Pr(inc)] > 75% are generally considered strong candidates for inclusion as 'key predictors' of the response variable (Thomson *et al*., 2007). Linear coefficients of correlation (r)

were determined using the Pearson's correlation coefficient and highly-correlated variables (r > 0.7) were excluded from this modelling because they influence the results of BMA analyses (Thomson *et al.*, 2007). Values of final candidate predictor variables used in generalized linear models are given in Table 2.

For all models assessed, the number of predictor variables was restricted to four (where N = 32-38) or seven (where N = 70) based on empirical evidence that the risk of over-fitting regression models is high if the number of predictors is more than approximately n/10 (Harrell *et al.*, 1984). Predictors were selected by initially running the model with all variables, and then rerunning the model with only the seven/four variables with the highest Pr(inc). Model averaging further reduces the potential risk of over-fitting because the probability that multiple models represent spurious response–predictor relationships is substantially lower than the probability that any single model is over-fitted (Thomson *et al*. 2007).

Using SOC stocks based on equivalent soil mass was explored but gave similar results, albeit with poorer models (R^2 = 0.01-0.40, N = 32-70) when compared with those using SOC stocks based on fixed depths (R^2 = 0.07-0.64, N = 32-70). Consequently, all analyses presented here are for SOC stocks based on fixed depths.

3. Results

3.1 Differences in SOC between land uses: individual site analysis

When considering the total dataset (N =117 sites), there was no statistically significant (P <0.05) difference in total SOC (TOC; data not shown) between land uses for the majority of sites (60%). Based on the assumption that any differences between land uses were attributable to the plantings (see Section 2.8), and when only sites with adequate replication (N = 71) were included, there were significant increases in TOC stocks (48% of sites), few significant decreases (4% of sites) or no significant difference (48% of sites; see Table A1). Significant differences in TOC stocks were more common at 0-10 cm depth than at 10-30 cm depth (Table A1), and in

previously-cropped than previously-grazed sites (data not shown). Across all 71 sites, the mean (±SE) rate of increase in TOC stocks was 0.57±0.06 Mg ha^{-1} y^{-1} at 0-30 cm depth (Table 3).

POC increased significantly at 46% of sites, compared with 27% of sites for the humus (HOC) fraction. HOC comprised the largest proportion of TOC (0.55±0.01 and 0.53±0.01), followed by ROC (0.25±0.01 and 0.25±0.01) and POC (0.19±0.01 and 0.22±0.01), for the agricultural pair/baseline and the plantings, respectively (data not shown). The largest relative difference was observed for the labile fraction, with a mean rate of increase in POC stock of 3.1±0.5 % y^{-1} compared with 1.2±0.3 % y^{-1} for HOC stocks (Table 3). Across the 71 sites, mean rates of increase in stocks of POC and HOC were 0.19±0.04 and 0.21±0.06 Mg ha^{-1} y^{-1}, respectively (Table 3).

3.2 Key factors influencing differences in SOC

3.2.1 Effects of previous land use

Mean annual difference in TOC stock was higher for former cropped than grazed land at 0-10 cm depth (df = 68, t = 2.09, P = 0.04; Fig. 2a) but not for 0-30 cm depth (df = 68, t = 1.06, P = 0.29). Conversely, TOC stocks in the agricultural pair or baseline were significantly lower for previously-cropped than previously-grazed sites both at 0-10 cm (df = 68, t = 2.48, P = 0.02) and 0-30 cm depth (df = 68, t = 2.33, P = 0.02; Fig. 2b). However, unlike TOC, there was no significant difference between previous land uses for mean annual difference in stocks of POC or HOC (df = 68, t < 0.80, P > 0.42).

3.2.2 Effects of multiple predictor variables across sites

For the previously-grazed sites, 'reference' TOC (i.e. TOC in the agricultural pair or baseline; Fig. 3a) and temperature were strong predictors of the difference in TOC at 0-10 cm depth, with differences becoming less pronounced on sites with higher reference (or initial) TOC (Table 4). When assessing the difference in TOC at 0-30 cm depth, 'reference' TOC remained a strong

predictor but the models were on average poor (R^2 = 0.24). For the previously-cropped sites, the strong predictors of difference in TOC were long-term mean annual rainfall and its interaction with planting age (Table 4). Differences in TOC increased with stand age, but with increases only being pronounced at sites where MAR was > 550 mm y^{-1} (Fig. 3b). The model for difference in TOC at 0-10 cm depth was similar to that at 0-30 cm, but was poor (R^2 = 0.14) at 10-30 cm depth with no strong predictors (Table 4).

Because the largest differences in SOC fractions were observed in the surface soil, only models for 0-10 cm are presented for differences in POC and HOC (Table 5). Across the previously-cropped sites, strong predictors of difference in POC and HOC in the surface soil included long-term MAR, minimum temperature and measures of stand composition including the proportions of acacia trees and nitrogen-fixing trees (Table 5). Long-term MAR, minimum temperature and the proportion of acacia trees had a positive influence, while the proportion of nitrogen fixing trees had a negative effect. By comparison, across the previously-grazed sites, 'reference' SOC was the only strong predictor of difference in POC while 'reference' SOC and climatic factors including long-term MAR and long-term maximum temperature were strong predictors of difference in HOC (Table 5). 'Reference' SOC had a negative influence on difference while long-term MAR and long-term maximum temperature had a positive effect.

3.3 Change in litter following reforestation

Across all 106 sites with litter data (53 existing, 53 new), total litter stocks in the plantings ranged from 2.4 to 30.6 Mg DM ha^{-1}, with rates of litter accumulation ranging from 0.22 to 2.27 Mg DM ha^{-1} y^{-1}. Similarly, across the 65 sites corresponding to sites with adequate sampling intensity for SOC, mean accumulation rates of litter dry matter and litter carbon were 0.94±0.06 Mg DM ha^{-1} y^{-1} and 0.39±0.02 Mg C ha^{-1} y^{-1}, respectively. When the combined change in soil and litter stock following reforestation was considered at the 65 sites with adequate sampling and

litter data, mean rates of accumulation were 1.25±0.20 and 0.79±0.12 Mg C ha^{-1} y^{-1} for previously-cropped (*N* = 30) and previously-grazed sites (*N* = 34), respectively (Fig. 2c).

4. Discussion

4.1 Change in SOC stocks following reforestation

Reforestation of agricultural land with environmental plantings resulted in a difference in stocks of total SOC (0-30 cm) of 0.57±0.06 Mg C ha^{-1} y^{-1} (Table 3). Significant differences in total SOC (sequestration) following reforestation were found at 52% of sites; for the majority of these total SOC increased, while total SOC decreased at 4% of sites (Table A1). Differences in total SOC were largely a result of differences in the POC fraction with significant increases at 48% of sites, compared only 27% of sites for the HOC fraction (Table A1). Relative increase was highest for POC but because HOC comprised the largest proportion of total SOC of all fractions, absolute differences in POC and HOC were similar (Table 3).

The increase in POC found here agrees with studies of afforestation of grassland across a rainfall gradient in South America (Berthrong *et al.*, 2012), and of cropland in Midwestern USA (DeGryze *et al.*, 2004). There are few comparable studies of HOC, as most report on 'recalcitrant C', which is equivalent to the combined humus and resistant fractions in our study. No change in recalcitrant C following reforestation was found in young deciduous *Populus* (DeGryze *et al.*, 2004) or evergreen *Eucalyptus*-*Acacia* plantings (Kasel *et al*., 2011). However, increases in stocks of recalcitrant SOC have been reported (Del Galdo *et al.*, 2003; Cunningham *et al.*, 2015b). Carbon in the humus fraction is considered to be more stabilised against microbial decomposition due to formation of organo-mineral complexes (Baldock and Skjemstad, 2000). By comparison, the resistant fraction is highly-carbonised organic material such as charcoal (Baldock and Skjemstad, 2000) which can survive for > 500 years (Lehmann *et al.*, 2008) and was therefore assumed not to change over the timeframe considered here.

There is little evidence for substantial change in soil carbon in the first three decades after reforestation of former pastures (e.g. Post and Kwon, 2000; Guo and Gifford, 2002; Paul *et al.*, 2002; Hoogmoed *et al.*, 2012; Prior *et al.*, 2015). Our survey showed no effect of planting age on SOC stocks, although differences ranged from increases, to negligible differences, to decreases at a small number (n = 4) of sites (Table A1). Similarly, there were no consistent changes in soil carbon following reforestation with environmental plantings aged up to 45 years (Cunningham *et al*., 2015b). Further, a review of reforestation studies, largely of *Eucalyptus* species in Australia, found little evidence for changes in soil carbon the first three decades (Hoogmoed *et al.*, 2012), but increases in SOC were observed in some younger (<10-year-old) plantings. These findings suggest that there may be site-specific factors influencing change in SOC following reforestation with environmental plantings. Further, which factors are important is likely to change depending on the scale (i.e. within regions e.g. Cunningham *et al*., 2015b, nationally e.g. this study, or globally e.g. Paul *et al*., 2002) at which changes in SOC are considered.

We believe the findings of our survey are robust because of adequate sampling and pairing of the plantings and adjacent fields. It is likely that many previous studies were not adequate in terms of sampling intensity (see Cunningham *et al.*, in review), so they were more likely to be inconclusive. We attributed any differences in SOC between the agricultural land use and the plantings at the paired sites to be due to reforestation based on the assumption that the agricultural land use is in steady state, i.e. management practices in the agricultural pair did not change after reforestation. We believe this assumption to be reasonable and, at least in part, verified by management histories in the 10 years prior to sampling collected for each of the agricultural pairs. Further, similar significant increases in TOC were found for 13 of the 17 sites with baseline and repeat sampling. With respect to pairing, recent detailed comparison of the mineralogy using the MIR spectra of soils from the individual paired sites used here has shown that the pairs share the same mineralogy, supporting the assumption that the sites were paired adequately with respect to soil type (Madhavan, *unpublished*). This, combined with the filtering

of the dataset to include only sites with adequate sampling intensity, gives support to the robustness of the observed changes in SOC.

4.2 Predictors of change in SOC following reforestation

4.2.1 Effects of previous land use

Previous land use had a significant effect on changes in soil carbon following reforestation (Fig. 2). SOC stocks are generally higher in grazed than cropped land uses at a specific location (e.g. Davy and Koen, 2013) and across regions (Rabbi *et al.*, 2014; Hobley *et al.*, 2015). At a specific location, this difference is largely related to the higher disturbance (from tillage) of soil under cropping compared with pasture soils, which have higher retention of carbon in aggregates and more inputs of plant residues from roots (Percival *et al.*, 2000). Across regions, cropped lands tend to be in areas of lower rainfall and, within these areas, we might expect increased SOC with increased rainfall, particularly when increased rainfall coincides with cooler temperatures. Here, the significantly lower SOC stocks in soils under cropping than grazing (Fig. 2b) was consistent with earlier surveys (e.g. Paul *et al*., 2002). Because cropped lands are most depleted in SOC in the surface soil, reforestation of previously-cropped land is likely to lead to larger increases at the surface. In contrast, the surface (and total profile) of grazed lands is less likely to be depleted (e.g. see Hobley *et al*., 2015), so the likelihood of soil carbon sequestration at these sites is lower. Our survey showed that the magnitude of change in SOC following reforestation was significantly higher for previously-cropped than previously-grazed sites in the 0-10 cm soil depth (Fig 2a). This agrees the findings of with existing meta-analyses (Laganiere *et al*., 2010; Guo and Gifford 2002; Paul *et al*. 2002), and suggests that sites with lower initial carbon (due to greater depletion) have a greater capacity to sequester carbon following reforestation.

4.2.2 Effects of multiple predictor variables across sites

The largest proportional increases in SOC following reforestation are generally found in lower rainfall areas (< 1200 mm y^{-1}; Guo and Gifford, 2002), such as those in this survey. Long-term mean annual rainfall was a strong predictor of difference in TOC for 0-10 and 0-30 cm depths for previously-cropped (but not previously-grazed) sites (Table 4), with a trend of increasing difference in TOC stocks with increasing rainfall for the 0-30 cm layer (Fig. 3b). Similarly, for previously-cropped (but not previously-grazed) sites, difference in POC was positively related to MAR in the 0-10 cm layer (Table 5). A survey of agricultural lands in eastern Australia also found that SOC in surface soil was driven predominantly by climate, particularly precipitation, vapour pressure deficit and relative humidity (Hobley *et al.*, 2015). Despite the rainfall effect at previously-cropped sites, other variables potentially affecting water availability, such as access to groundwater or landscape position, generally were not strong predictors of change in SOC stocks (Tables 4 and 5).

Difference in TOC in the surface soil was related negatively to SOC in the agricultural soil on previously-grazed sites, but not for previously-cropped sites (Table 4; Fig 3a). This suggests that reforestation of more productive pasture may lead to a loss in SOC. More productive pastures are likely to have higher initial SOC stocks, which have the potential to increase losses in SOC stocks observed in early decades after reforestation (Paul *et al*., 2002). Soils with high carbon stocks may have higher nitrogen stocks and differing microbial communities (e.g. Fierer *et al.*, 2009), which play a key role in organic matter decomposition. Reports of the effects of reforestation by environmental plantings on soil microbial communities are limited, but a recent study of plantings on former pastures in relation to nitrogen supply found that differences in microbial communities were specific to the tree species planted (Hoogmoed *et al*., 2014).

The species of trees planted are important to soil carbon sequestration following reforestation, with the largest increases under broadleaf species (27%), intermediate responses under eucalypts

and pines (12%) and little change under other conifers (2%, Laganière *et al.*, 2010). Plantings that include nitrogen-fixing tree species can have higher productivity (Forrester *et al.*, 2006) and higher retention of original soil carbon stocks (Resh *et al.*, 2002), suggesting their inclusion may accelerate accumulation of SOC (Cunningham *et al*., 2015a). Here, there were some species effects on differences in HOC across the previously-cropped sites, with significant predictors including the proportion of acacia trees and proportion of N-fixing trees (Table 5). The proportion of acacia trees had a positive influence on change in HOC fractions in the surface 10 cm, suggesting that inclusion of these in plantings may increase sequestration of more stable forms of carbon on previously-cropped sites. Counter-intuitively, differences in these fractions were negatively related to the proportion of nitrogen fixing trees in the planting (Table 5). Similarly, previous studies in environmental plantings comparing changes in SOC between fixers and non-fixer trees found that effects were restricted to specific *Acacia* species (Hoogmoed *et al*., 2014; Kasel *et al*., 2011).

4.3 Changes in litter stocks following reforestation

Total litter stocks under young (≤ 29-year-old) environmental plantings ranged from 2.4 to 30.6 Mg DM ha^{-1}. This is comparable with reported litter stocks, which range from 4.1-24.6 Mg DM ha^{-1} in eucalypt-dominated open forests and woodlands (Pressland, 1982; Hutson, 1985; Adams and Attiwill, 1991; Crockford and Richardson, 1998) and 19-34 Mg DM ha^{-1} under 26-year-old farm forestry plantations of eucalypts (Harper *et al*., 2012) across similar rainfalls. Accumulation of litter C varied widely among sites, with the mean (±SE) rate of accumulation of 0.39±0.02 Mg C ha^{-1} y^{-1}, which was similar to the rate in soil (0.57±0.06 Mg C ha^{-1} y^{-1}). When the additional changes in litter C were added to the changes in soil carbon, rates of C sequestration were in the order of 1.25±0.20 and 0.79±0.12 Mg C ha^{-1} y^{-1} for previously-cropped and previously-grazed sites, respectively (Fig. 2d).

4.4 Conclusion

There is potential to increase SOC stocks following reforestation with mixed-species environmental plantings in the first two decades. Reforestation on former cropping sites has a higher capacity to sequester SOC, at least in the surface soil, than on former grazing sites. Differences in SOC stocks were generally predicted weakly by empirical models, suggesting that either the factors driving these changes may be largely site-specific, or that predictors not included here are more important. For example, there is potential for other processes, such as increased deposition and reduced erosion, to increase site-based SOC in these agricultural landscapes (Bird *et al.*, 1992).

Factors that were correlated strongly with difference in SOC stocks were predominantly long-term rainfall at previously-cropped sites, and SOC stocks in the agricultural pair/baseline and temperature at previously-grazed sites. This improved understanding of the potential predictors of sequestration of soil carbon following reforestation across a broad range of site conditions and management practices suggests that rates could be improved by planting on previously-cropped sites in areas where rainfall is > 550 mm yr^{-1}. However, this does not account for the rates of carbon sequestration in above-ground biomass and debris, and the current focus of reforestation on previously-grazed sites, presumably due to demand for prime cropping land. More targeted sampling to test specific hypotheses and drivers, such as addressing differences between former cropped and grazed land uses, particularly in older plantings (> 30 years old) is the important next step in building accurate models of carbon sequestration in soil following reforestation. Experimental plantings are required to investigate the effects of species mix and soil type on soil organic carbon following reforestation with mixed-species environmental plantings.

Acknowledgements

This work was largely funded by the Australian Government's Filling the Research Gap Program and CSIRO, with additional financial support from Victorian Department of Environment, Land, Water and Planning and Central Tablelands Local Land Council. For assistance with field sampling/processing samples we thank Amanda Schapel (two SA sites), Micah Davies, Gordon McLachlan, John Larmour, Melanie Bullock and Pandora Holliday (15 NSW sites), Simon Murphy, Tarek Murshed, Tom Fairman, Rob Law, Ben Smith and Gabi Szegedy (17 Victorian sites, including the 10 new riparian sites), Jarrod Hodgson and Phil De Zylva (five Victorian sites and two NSW sites, including the three intensive sites), and Tim Morald and Andrew Wherrett (2 WA sites). We also thank Debbie Crawford for assistance with preparation of Figure 1, John Larmour for completing the new inventories for biomass estimation, Hamish Luck for assistance with compiling information from FullCAM, Michael Davy and Geoff Minchin for assistance with locating NSW field sites, John Raison for initial advice on sampling design, landholders for permission to access their properties, and Nellie Hobley and Daniel Mendham for providing comments that improved the manuscript. We wish to thank the Australian Research Council for supporting SCC from an ARC Linkage grant (LP0990038), TRC via the award of a Future Fellowship (FT120100463) and MPP through an ARC Laureate Fellowship to Richard J. Hobbs.

References

ABARES, 2004. Atlas of Australian Soils. Australian Bureau of Agricultural and Resource Economics and Sciences, Department of Agriculture, Fisheries and Forestry, Canberra.

Adams, M.A., Attiwill, P.M., 1991. Nutrient balance in forests of northern Tasmania. 1. Atmospheric inputs and within-stand cycles. Forest Ecology and Management 44, 93-113.

Baldock, J.A., Hawke, B., Sanderman, J., Macdonald, L.M., 2013a. Predicting contents of carbon and its component fractions in Australian soils from diffuse reflectance mid-infrared spectra. Soil Research 51, 577-583.

Baldock, J.A., Sanderman, J., Macdonald, L.M., Puccini, A., Hawke, B., Szarvas, S., McGowan, J., 2013b. Quantifying the allocation of soil organic carbon to biologically significant fractions. Soil Research 51, 561-576.

Baldock, J.A., Skjemstad, J.O., 2000. Role of the soil matrix and minerals in protecting natural organic materials against biological attack. Organic Geochemistry 31, 697-710.

Berthrong, S.T., Pineiro, G., Jobbagy, E.G., Jackson, R.B., 2012. Soil C and N changes with afforestation of grasslands across gradients of precipitation and plantation age. Ecological Applications 22, 76-86.

Bird, P.R., Bicknell, D., Bulman, P.A., Burke, S.J.A., Leys, J.F., Parker, J.N., Vandersommen, F.J., Voller, P., 1992. The role of shelter in Australia for protecting soils, plants and livestock. Agroforestry Systems 20, 59-86.

Canadell, J.G., Raupach, M.R., 2008. Managing forests for climate change mitigation. Science 320, 1456-1457.

Crockford, R., Richardson, D., 1998. Litterfall, litter and associated chemistry in a dry sclerophyll eucalypt forest and a pine plantation in south-eastern Australia: 1. Litterfall and litter. Hydrological Processes 12, 365-384.

Crossman, N.D., Bryan, B.A., Summers, D.M., 2011. Carbon Payments and Low-Cost Conservation. Conservation Biology 25, 835-845.

Cunningham, S., Mac Nally, R., Baker, P., Cavagnaro, T., Beringer, J., Thomson, J., Thompson, R., 2015a. Balancing the environmental benefits of reforestation in agricultural regions. Perspectives in Plant Ecology, Evolution and Systematics 17, 301-317.

Cunningham, S., Roxburgh, S., Cavagnaro, T., Paul, K., in review. A robust method for designing efficient sampling of environmental variables using environmental surfaces to design efficient sampling protocols. Ecological Indicators?, ?

Cunningham, S.C., Cavagnaro, T.R., Mac Nally, R., Paul, K.I., Baker, P.J., Beringer, J., Thomson, J.R., Thompson, R.M., 2015b. Reforestation with native mixed-species plantings in

a temperate continental climate effectively sequesters and stabilizes carbon within decades. Global Change Biology 21, 1552-1566.

Cunningham, S.C., Metzeling, K.J., Mac Nally, R., Thomson, J.R., Cavagnaro, T.R., 2012. Changes in soil carbon of pastures after afforestation with mixed species: Sampling, heterogeneity and surrogates. Agriculture Ecosystems & Environment 158, 58-65.

Davy, M.C., Koen, T.B., 2013. Variations in soil organic carbon for two soil types and six land uses in the Murray Catchment, New South Wales, Australia. Soil Research 51, 631-644.

DeGryze, S., Six, J., Paustian, K., Morris, S.J., Paul, E.A., Merckx, R., 2004. Soil organic carbon pool changes following land-use conversions. Global Change Biology 10, 1120-1132.

Del Galdo, I., Six, J., Peressotti, A., Cotrufo, M.F., 2003. Assessing the impact of land-use change on soil C sequestration in agricultural soils by means of organic matter fractionation and stable C isotopes. Global Change Biology 9, 1204-1213.

DIICCSRTE, 2014. Australian National Greenhouse Accounts. National Inventory Report 2011, Volume 2. The Australian Government submission to the United Nations Convention of Climate Change, April 2013. The Department of Industry, Innovation, Climate Change, Science, Research and Tertiary Education (DIICCSRTE), Canberra, Australia, p. 154.

Fernandez-Nunez, E., Rigueiro-Rodriguez, A., Mosquera-Losada, M.R., 2010. Carbon allocation dynamics one decade after afforestation with *Pinus radiata* D. Don and *Betula alba* L. under two stand densities in NW Spain. Ecological Engineering 36, 876-890.

Fierer, N., Strickland, M.S., Liptzin, D., Bradford, M.A., Cleveland, C.C., 2009. Global patterns in belowground communities. Ecology Letters 12, 1238-1249.

Forrester, D.I., Pares, A., O'Hara, C., Khanna, P.K., Bauhus, J., 2013. Soil organic carbon is increased in mixed-species plantations of *Eucalyptus* and nitrogen-fixing *Acacia*. Ecosystems 16, 123-132.

Guo, L.B., Gifford, R.M., 2002. Soil carbon stocks and land use change: a meta analysis. Global Change Biology 8, 345-360.

Harper, R.J., Okom, A.E.A., Stilwell, A.T., Tibbett, M., Dean, C., George, S.J., Sochacki, S.J., Mitchell, C.D., Mann, S.S., Dods, K., 2012. Reforesting degraded agricultural landscapes with Eucalypts: Effects on carbon storage and soil fertility after 26 years. Agriculture Ecosystems & Environment 163, 3-13.

Harrell, F., Lee, K., Califf, R., Pryor, D., Rosati, R., 1984. Regression modelling strategies for improved prognostic prediction. Statistics in Medicine 3, 143–152.

Heanes, D., 1984. Determination of total organic-c in soils by an improved chromic acid digestion and spectrophotometric procedure. Communications in Soil Science and Plant Analysis 15, 1191-1213.

Hobley, E., Wilson, B., Wilkie, A., Gray, J., Koen, T., 2015. Drivers of soil organic carbon storage and vertical distribution in Eastern Australia. Plant and Soil 390, 111-127.

Hoogmoed, M., Cunningham, S.C., Baker, P.J., Beringer, J., Cavagnaro, T.R., 2014. Is there more soil carbon under nitrogen-fixing trees than under non-nitrogen-fixing trees in mixed-species restoration plantings? Agriculture Ecosystems & Environment 188, 80-84.

Hoogmoed, M., Cunningham, S.C., Thomson, J.R., Baker, P.J., Beringer, J., Cavagnaro, T.R., 2012. Does afforestation of pastures increase sequestration of soil carbon in Mediterranean climates? Agriculture Ecosystems & Environment 159, 176-183.

Hulvey, K.B., Hobbs, R.J., Standish, R.J., Lindenmayer, D.B., Lach, L., Perring, M.P., 2013. Benefits of tree mixes in carbon plantings. Nature Climate Change 3, 869-874.

Hutson, B.R., 1985. Rates of litterfall and organic matter turnover at three South Australian indigenous forest sites. Australian Journal of Ecology 10, 351-359.

Jeffrey, S.J., Carter, J.O., Moodie, K.M., Beswick, A.R., 2001. Using spatial interpolation to construct a comprehensive archive of Australian climate data. Environmental Modelling and Software 16, 309-330.

Kasel, S., Singh, S., Sanders, G.J., Bennett, L.T., 2011. Species-specific effects of native trees on soil organic carbon in biodiverse plantings across north-central Victoria, Australia. Geoderma 161, 95-106.

Laganière, J., Angers, D.A., Pare, D., 2010. Carbon accumulation in agricultural soils after afforestation: a meta-analysis. Global Change Biology 16, 439-453.

Lal, R., 2005. Forest soils and carbon sequestration. Forest Ecology and Management 220, 242-258.

Lehmann, J., Skjemstad, J., Sohi, S., Carter, J., Barson, M., Falloon, P., Coleman, K., Woodbury, P., Krull, E., 2008. Australian climate-carbon cycle feedback reduced by soil black carbon. Nature Geoscience 1, 832-835.

Lima, A.M.N., Silva, I.R., Neves, J.C.L., Novais, R.F., Barros, N.F., Mendonca, E.S., Smyth, T.J., Moreira, M.S., Leite, F.P., 2006. Soil organic carbon dynamics following afforestation of degraded pastures with *Eucalyptus* in southeastern Brazil. Forest Ecology and Management 235, 219-231.

Madhavan, D., Baldock, J., Read, Z., SC, M., Cunningham, S., Perring, M., Herrmann, T., Lewis, T., Cavagnaro, T., England, J., Paul, K., Weston, C., Baker, T., 2016. Mid-infrared spectroscopic techniques for estimating soil organic carbon fractions under environmental plantings and agricultural land uses. In preparation.

McKenzie, N., Jacquier, D., Ashton, L., Cresswell, H., 2000. Estimation of Soil Properties Using the Atlas of Australian Soils. CSIRO Land and Water Technical Report 11/00. CSIRO Land and Water, Canberra.

Mitchell, C.D., Harper, R.J., Keenan, R.J., 2012. Current status and future prospects for carbon forestry in Australia. Australian Forestry 75, 200-212.

Parfitt, R.L., Percival, H.J., Dahlgren, R.A., Hill, L.F., 1997. Soil and solution chemistry under pasture and radiata pine in New Zealand. Plant and Soil 191, 279-290.

Paul, K.I., Jacobsen, K., Koul, V., Leppert, P., Smith, J., 2008. Predicting growth and sequestration of carbon by plantations growing in regions of low-rainfall in southern Australia. Forest Ecology and Management 254, 205-216.

Paul, K.I., Polglase, P.J., 2004. Prediction of decomposition of litter under eucalypts and pines using the FullCAM model. Forest Ecology and Management 191, 73-92.

Paul, K.I., Polglase, P.J., Nyakuengama, J.G., Khanna, P.K., 2002. Change in soil carbon following afforestation. Forest Ecology and Management 168, 241-257.

Paul, K.I., Roxburgh, S.H., England, J.R., de Ligt, R., Larmour, J.S., Brooksbank, K., Murphy, S., Ritson, P., Hobbs, T., Lewis, T., Preece, N.D., Cunningham, S.C., Read, Z., Clifford, D., Raison, R.J., 2015. Improved models for estimating temporal changes in carbon sequestration in above-ground biomass of mixed-species environmental plantings. Forest Ecology and Management 338, 208-218.

Paul, K.I., Roxburgh, S.H., England, J.R., Ritson, P., Hobbs, T., Brooksbank, K., Raison, R.J., Larmour, J.S., Murphy, S., Norris, J., Neumann, C., Lewis, T., Jonson, J., Carter, J.L., McArthur, G., Barton, C., Rose, B., 2013. Development and testing of generic allometric equations for estimating above-ground biomass in mixed-species environmental plantings. Forest Ecology and Management 310, 483-494.

Paul, K.L., Polglase, P.J., Richards, G.P., 2003. Predicted change in soil carbon following afforestation or reforestation, and analysis of controlling factors by linking a C accounting model (CAMFor) to models of forest growth (3PG), litter decomposition (GENDEC) and soil C turnover (RothC). Forest Ecology and Management 177, 485-501.

Paz-Gonzalez, A., Vieira, S.R., Castro, M.T.T., 2000. The effect of cultivation on the spatial variability of selected properties of an umbric horizon. Geoderma 97, 273-292.

Percival, H.J., Parfitt, R.L., Scott, N.A., 2000. Factors controlling soil carbon levels in New Zealand grasslands: Is clay content important? Soil Science Society of America Journal 64, 1623-1630.

Piotto, D., 2008. A meta-analysis comparing tree growth in monocultures and mixed plantations. Forest Ecology and Management 255, 781-786.

Polglase, P.J., Reeson, A., Hawkins, C.S., Paul, K.I., Siggins, A.W., Turner, J., Crawford, D.F., Jovanovic, T., Hobbs, T.J., Opie, K., Carwardine, J., Almeida, A., 2013. Potential for forest carbon plantings to offset greenhouse emissions in Australia: economics and constraints to implementation. Climatic Change 121, 161-175.

Post, W.M., Kwon, K.C., 2000. Soil carbon sequestration and land-use change: processes and potential. Global Change Biology 6, 317–327.

Prescott, C.E., 2010. Litter decomposition: what controls it and how can we alter it to sequester more carbon in forest soils? Biogeochemistry 101, 133-149.

Pressland, A.J., 1982. Litter production and decomposition from an overstorey of *Eucalyptus* species on two catchments in the New England region of New South Wales. Australian Journal of Ecology 7, 171-180.

Prior, L., Paul, K., Davidson, N., Hovenden, M., Nichols, S., Bowman, D., 2015. Evaluating carbon storage in restoration plantings in the Tasmanian Midlands, a highly modified agricultural landscape. The Rangeland Journal, In press.

Raftery, A., Hoeting, J., Volinsky, C., Painter, I., Yeung, K.Y. (Eds.), 2008. The BMA package: version 3.05., http://cran.r-project.org/web/packages/BMA/BMA.pdf.

Read, Z., 2015. Soil organic carbon sequestration following land use change: Two case studies. PhD Thesis, Fenner School of Environment and Society. Australian National University, Canberra, Australia.

Resh, S.C., Binkley, D., Parrotta, J.A., 2002. Greater soil carbon sequestration under nitrogen-fixing trees compared with *Eucalyptus* species. Ecosystems 5, 217-231.

Silver, W.L., Ostertag, R., Lugo, A.E., 2000. The potential for carbon sequestration through reforestation of abandoned tropical agricultural and pasture lands. Restoration Ecology 8, 394-407.

Skjemstad, J., Spouncer, L., 2003. Integrated soils modelling for the National Carbon Accounting System. National Carbon Accounting System Technical Report No. 36. Australian Greenhouse Office, Canberra.

Skjemstad, J.O., Spouncer, L.R., Cowie, B., Swift, R.S., 2004. Calibration of the Rothamsted organic carbon turnover model (RothC ver. 26.3), using measurable soil organic carbon pools. Australian Journal of Soil Research 42, 79-88.

Spain, A.V., 1984. Litterfall and the standing crop of litter in three tropical Australian rainforests. Journal of Ecology 72, 947-961.

Specht, A., West, P.W., 2003. Estimation of biomass and sequestered carbon on farm forest plantations in northern New South Wales, Australia. Biomass & Bioenergy 25, 363-379.

Thomson, J.R., Mac Nally, R., Fleishman, E., Horrocks, G., 2007. Predicting bird species distributions in reconstructed landscapes. Conservation Biology 21, 752-766.

Turner, J., 1986. Organic matter accumulation in a series of *Eucalyptus grandis* plantations. Forest Ecology and Management 17, 231-242.

Turner, J., Lambert, M., 2000. Change in organic carbon in forest plantation soils in eastern Australia. Forest Ecology and Management 133, 231-247.

VandenBygaart, A.J., 2006. Monitoring soil organic carbon stock changes in agricultural landscapes: Issues and a proposed approach. Canadian Journal of Soil Science 86, 451-463.

Wendt, J.W., Hauser, S., 2013. An equivalent soil mass procedure for monitoring soil organic carbon in multiple soil layers. European Journal of Soil Science 64, 58-65.

Wilson, B.R., Moffatt, A.J., Nortcliff, S., 1997. The nature of three ancient woodland soils in southern England. Journal of Biogeography 24, 633-646.

List of Figures

Figure 1 Location of the 117 sites where soils and litter were sampled under mixed-species environmental plantings. Grey shading shows regions where previously-developed calibrations for aboveground biomass of environmental plantings (Paul *et al*., 2015) can be applied.

Figure 2 Comparison between previously-cropped (N = 32) and previously-grazed (N = 38) sites for: a) mean annual difference in total organic carbon (ΔTOC 0-10 cm) stocks following reforestation in the surface soil, b) total organic carbon (TOC) stocks in the agricultural pair or baseline (0-30 cm), and c) mean annual difference in combined soil and litter C stocks (ΔTOC 0-30 cm + litter C) following reforestation (0-30 cm). Error bars are standard error of the mean.

Figure 3 Scatterplots showing relationships between a) difference in total organic carbon (ΔTOC) stocks in the surface 10 cm and soil total C in the agricultural pair or baseline (Ag TOC) at the previously-grazed sites (N = 38), and b) ΔTOC to 30 cm depth and long-term mean annual rainfall (MAR) at the previously-cropped sites (N = 32).

Table 1 Summary of the coverage and source of the datasets collated, including sampling method (paired = paired with agricultural land use, repeat = repeat sampling from agricultural baseline), age, previous land use (G = grazing, C = cropping, including rotational cropping and grazing), productivity of the plantings (above-ground biomass (AGB) increment; low = < 4 Mg DM ha^{-1} y^{-1}, high = ≥ 4 Mg DM ha^{-1} y^{-1})[1] and soil texture (light = ≤ 20% clay, heavy = > 20% clay)[2].

Type	SOC	Litter	Sampling	Age	Previous land use		AGB increment		Soil texture		Source
	(*n* sites)	(*n* sites)		(y)	(*n* sites)		(*n* sites)		(*n* sites)		
					G	C	Low	High	Light	Heavy	
Existing	20	11	paired	1-19	20	-	11	9	5	15	Read (2015)
Existing	35	35	paired	5-45	35	-	20	15	30	5	Cunningham *et al*. (2015b)
Existing	2	-	paired	15-20	-	2	2	-	2	-	T. Herrmann, *pers. comm.*
Existing	7	7	paired	8-16	5	2	2	5	7	-	T. Baker, *pers. comm.*
Existing	17	-	baseline	8-20	3	14	14	3	-	17	B. Wilson, *pers. comm.*
New	53	53	paired, repeat[5]	6-29	28	28	33	23	15	41	This study
Total	**117**[6]	**106**		**1-45**	**85**	**32**	**66**	**51**	**58**	**59**	

[1] Based on Paul *et al*., (2015);

[2] Based on spatial layers underpinning the National Carbon Accounting Toolbox (DIICCSRTE 2014);

[3] Repeat sampling of 17 baseline sites (B. Wilson);

[4] Totals exclude baseline sampling.

Table 2 Variables used as candidate predictors of difference in soil organic carbon stocks and litter stocks following reforestation with mixed-species plantings (*N* = 117 sites).

Code	Mean	SD	Min	Max	Definition
Climate & water availability					
MAR[a]	647	135	380	1147	Long-term mean annual rainfall (mm)
PLEVAP[a]	1467	228	1063	2681	Mean annual evaporation since planting (mm)
MAXT[a]	21.1	1.9	17.0	25.6	Long-term mean monthly max. temp. (°C)
MINT[a]	8.5	1.3	5.9	11.5	Long-term mean monthly min. temp. (°C)
PLMAXTHOT[a]	31.2	2.3	23.9	35.3	Mean max. temp. of the hottest month since planting (°C)
PLMINTCOLD[a]	2.6	1.3	-0.3	6.0	Mean min. temp. of the coldest month since planting (°C)
ELEV[a]	372	209	41	1055	Elevation (m)
RIPARIAN[b]	-	-	-	-	Riparian landscape position
UPPER[b]	-	-	-	-	Upper landscape position
ADDWATER[b]	-	-	-	-	Access to additional water
Soils					
AGSOILTC0-30[c]	46.5	20.2	15.4	160.2	Total soil carbon content of agricultural pair, 0-30 cm depth (Mg ha^{-1})
AGSOILTC0-10[c]	25.7	10.9	8.9	90.4	Total soil carbon content of agricultural pair, 0-10 cm depth (Mg ha^{-1})
AGSOILTC10-30[c]	21.3	11.2	6.5	69.8	Total soil carbon content of agricultural pair, 10-30 cm depth (Mg ha^{-1})
CLAY[d]	26.6	10.0	5.0	53.0	Soil clay content (%)
TEXTURE[e]	2.9	1.0	1.0	5.0	Soil texture (1-5)
SODOSOL[b,e]	-	-	-	-	Sodosol soil type
Previous land use and age					
GRAZED[b]	-	-	-	-	Previous land use grazing
CROPPED[b]	-	-	-	-	Previous land use cropping
AGE	14	7	1	45	Planting age at sampling (y)
Management					
TS[b]	-	-	-	-	Planting established using tubestock
WIDTH	64	63	7	416	Mean planting width (m)
DENSITY[c]	1951	3269	124	20768	Density of trees and shrubs (ha^{-1})
PROPEUC[c]	0.53	0.25	0.00	1.00	Proportion of eucalypts
PROPTREE[c]	0.75	0.20	0.10	1.00	Proportion of trees
PROPACA[c]	0.31	0.25	0.00	1.00	Proportion of acacias (trees and shrubs)
PROPACATREE[c]	0.16	0.18	0.00	0.83	Proportion of acacia trees
PROPFIXER[c]	0.21	0.18	0.00	0.83	Proportion of N fixing trees
LIVEAGB[c]	62.8	57.3	0.4	411.5	Live aboveground biomass (Mg DM ha^{-1})
LAT[f]	-35.45	1.78	-38.67	-30.86	Longitude (° E)
LONG[f]	146.1	4.5	117.4	150.3	Latitude (° S)

[a] Derived from SILO data drill 1968-2012 (Jeffrey et al. 2001; https://www.longpaddock.qld.gov.au/silo/); [b] Binary categorical variable with value of 0 or 1; [c] Derived from plot measurements in this study; [d] Derived from FullCAM (DIICCSRTE 2014); [e] Derived from GIS layers in the Australian Soil Classification system of the Atlas of Australian Soils (ABARES 2004) and textural classes (McKenzie et al. 2000); [f] Geographic coordinates not used in main models.

Table 3 Difference in soil organic carbon (ΔSOC) following reforestation with mixed-species plantings at sites with adequate sampling intensity (*N* = 71, plantings aged 2-29 y). ΔSOC values are mean annual percentage change (% y^{-1}) and mean annual change (Mg C ha^{-1} y^{-1}); standard errors of the means are given in parentheses.

SOC fraction	**Depth (cm)**	**ΔSOC** % y^{-1}	Mg ha^{-1} y^{-1}
Total	0-30	1.5 (0.3)	0.57 (0.10)
(TOC)	0-10	1.4 (0.3)	0.21 (0.06)
	10-30	1.8 (0.5)	0.35 (0.10)
Particulate	0-30	3.1 (0.5)	0.19 (0.04)
(POC)	0-10	3.0 (0.5)	0.09 (0.02)
	10-30	4.9 (1.4)	0.09 (0.02)
Humus	0-30	1.2 (0.3)	0.21 (0.06)
(HOC)	0-10	1.1 (0.3)	0.09 (0.03)
	10-30	1.5 (0.5)	0.12 (0.04)

Table 4 Results of generalized linear modelling with Bayesian model averaging of difference in total organic carbon (ΔTOC) for fixed depths (0-10, 10-30, 0-30 cm) in relation to stand, management, climate and soil variables at sites with adequate sampling intensity, by previous land use. Only strong predictors Pr(inc) > 0.75 are shown, with the exception of predictors included with interactions (italicized). Predictor variable codes are defined in Table 2.

Response variable	***n***	**Mean model R^2**	**Predictor variable**	**Pr(inc)**	**Std. coefficient ± SD**
Grazed					
ΔTOC, 0-30 cm	38	0.24	AGSOILTC0-30	0.76	-36.9 ± 41.3
ΔTOC, 0-10 cm	38	0.46	AGSOILTC0-10	1.00	-15.3 ± 3.8
			MINT	0.79	10.8 ± 7.4
ΔTOC, 10-30 cm	38	0.20	-	-	-
Cropped					
ΔTOC, 0-30 cm	32	0.52	MAR	1.00	100.1 ± 41.7
			MAR×AGE	0.95	-94.4 ± 60.8
			AGE	*0.71*	*49.6 ± 47.8*
ΔTOC, 0-10 cm	32	0.53	MAR	1.00	72.3 ± 30.6
			MAR×AGE	0.94	-65.0 ± 44.6
			AGE	*0.68*	*33.5 ± 35.0*
ΔTOC, 10-30 cm	32	0.14	-	-	-

Table 5 Results of generalized linear modelling using Bayesian model averaging of difference in particulate (ΔPOC) and humus (ΔHOC) soil organic carbon fractions in the surface soil (0-10 cm) in relation to stand, management, climate and soil variables at sites with adequate sampling intensity by previous land use. Only strong predictors Pr(inc) > 75 shown. Predictor variables are defined in Table 2.

Response variable	***n***	**Mean model r^2**	**Predictor variable**	**Pr(inc)**	**Std. coefficient ± SD**
Previously-grazed					
ΔPOC	38	0.19	AGSOILTC0-10	0.90	-3.95 ± 2.01
ΔHOC	38	0.49	MAXT	1.00	25.56 ± 8.92
			MAR	1.00	10.46 ± 3.46
			AGSOILTC0-10	0.93	-5.92 ± 2.76
Previously-cropped					
ΔPOC	32	0.39	MAR	1.00	7.22 ± 2.06
ΔHOC	32	0.64	PROPACATREE	1.00	7.28 ± 2.23
			PROPNFIXTREE	1.00	-13.22 ± 2.69
			MINT	1.00	22.01 ± 6.95
			MAR	0.80	8.20 ± 5.49

Appendix 1

Table Differences in soil organic carbon (ΔSOC) following reforestation with mixed-species plantings at sites with adequate sampling intensity (*N* = 71). ΔSOC values are presented as mean annual percentage change (% y^{-1}) and mean annual change (Mg C ha^{-1} y^{-1}) for significant increases (+), significant decreases (-) and no significant change (0). Standard errors of the means are given in parentheses. POC = particulate organic carbon, HOC = humus organic carbon.

SOC fraction	**Depth (cm)**	**Change**	**ΔSOC** % y^{-1}	**ΔSOC** Mg ha^{-1} y^{-1}	***n***	**Age range** y
Total	0-30	+	2.7 (0.4)	1.1 (0.2)	34	2-23
		0	-	-	34	6-29
		-	-1.3 (0.2)	-0.9 (0.2)	3	9-18
	0-10	+	3.5 (0.5)	0.6 (0.1)	29	6-29
		0	-	-	36	2-21
		-	-1.4 (0.4)	-0.7 (0.4)	6	9-13
	10-30	+	4.3 (1.2)	0.9 (0.2)	25	2-23
		0	-	-	44	3-21
		-	-2.9 (1.2)	-0.3 (0.2)	2	18-29
POC	0-30	+	5.6 (0.8)	0.4 (0.1)	33	2-29
		0	-	-	33	3-21
		-	-1.8 (0.4)	-0.2 (0.1)	5	9-16
	0-10	+	5.5 (0.8)	0.2 (0.0)	28	8-29
		0	-	-	37	2-21
		-	-1.5 (0.2)	-0.2 (0.1)	6	9-16
	10-30	+	11.5 (2.0)	0.3 (0.1)	17	3-23
		0	-	-	46	2-21
		-	-5.1 (1.2)	-0.1 (0.0)	8	9-29
HOC	0-30	+	2.5 (0.3)	0.5 (0.1)	19	8-23
		0	-	-	47	2-29
		-	-1.3 (0.1)	-0.5 (0.1)	5	9-18
	0-10	+	4.0 (0.6)	0.3 (0.0)	20	8-29
		0	-	-	41	2-21
		-	-1.2 (0.2)	-0.2 (0.0)	10	9-18
	10-30	+	3.1 (0.4)	0.4 (0.1)	15	6-23
		0	-	-	48	2-21
		-	-1.8 (0.2)	-0.2 (0.0)	10	9-29

Figure 1

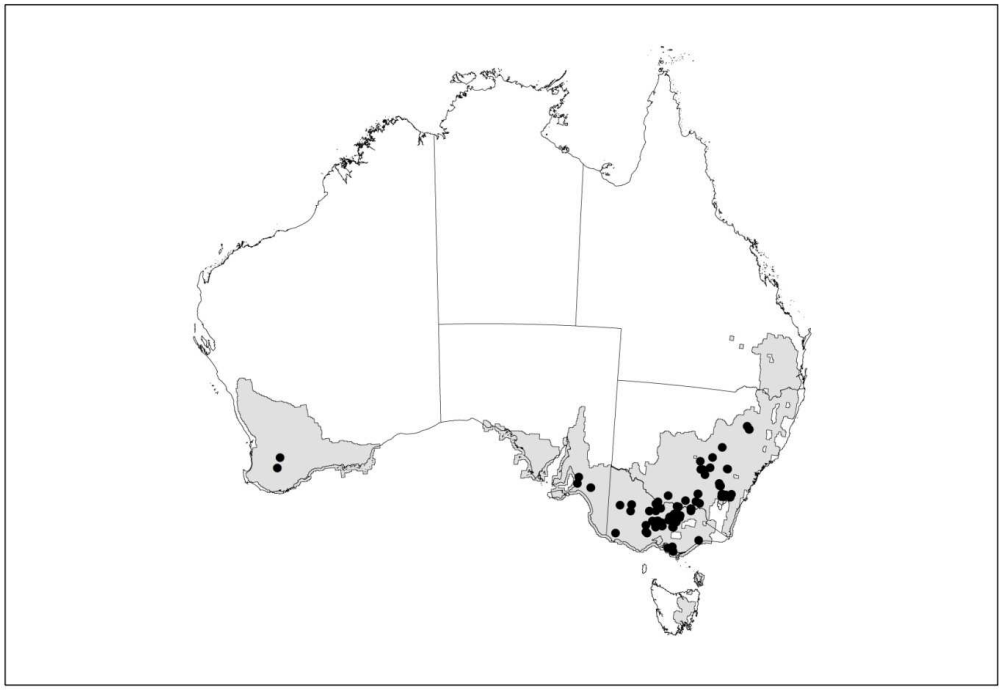


Figure 2

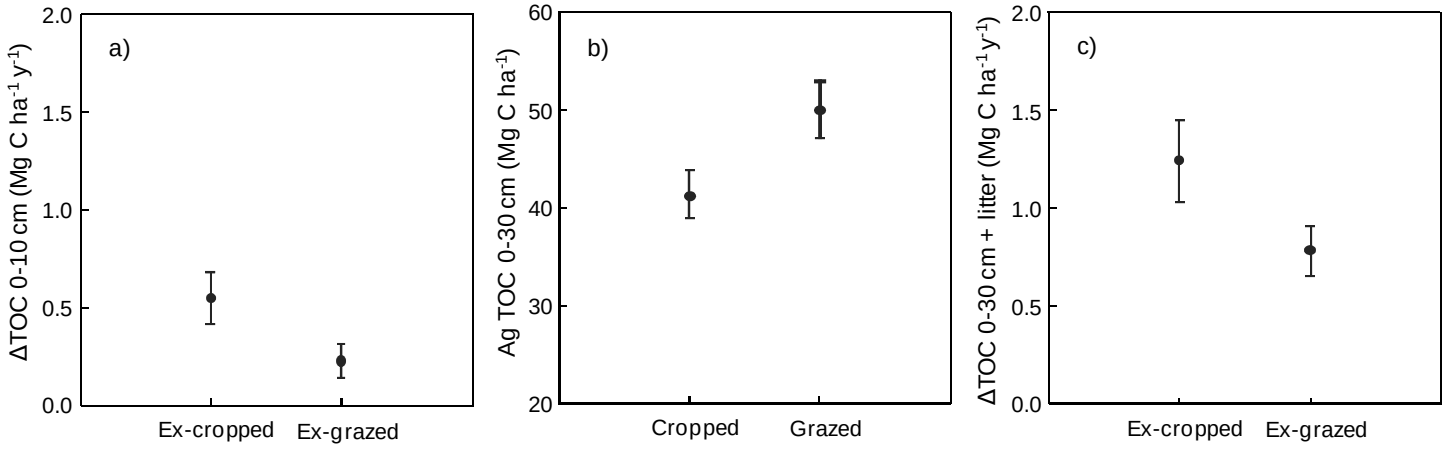
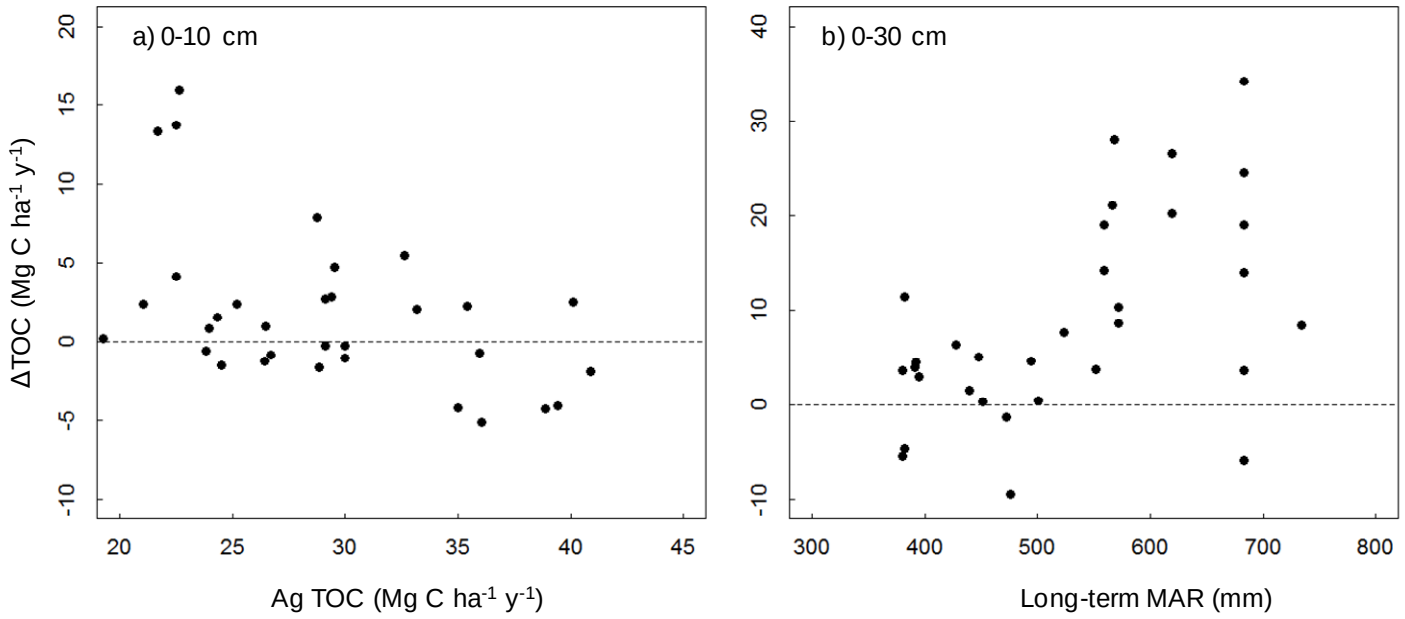

a)
ΔTOC 0-10 cm (Mg C ha-1 y-1)
2.0
1.5
1.0
0.5
0.0
Ex-cropped
Ex-grazed
b)
Ag TOC 0-30 cm (Mg C ha-1)
60
50
40
30
20
Cropped
Grazed
c)
ΔTOC 0-30 cm + litter (Mg C ha-1 y-1)
2.0
1.5
1.0
0.5
0.0
Ex-cropped
Ex-grazed

Figure 3


a) 0-10 cm
b) 0-30 cm
ΔTOC (Mg C ha-1 y-1)
Ag TOC (Mg C ha-1 y-1)
Long-term MAR (mm)
20
15
10
5
0
-5
-10
25
30
35
40
45
300
400
500
600
700
800