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Conversion of sub-tropical native vegetation to introduced conifer forest: Impacts on below-ground and above-ground carbon pools

Tom Lewis [a,b,*], Timothy E. Smith [a,b], Bruce Hogg [a], Scott Swift [a], Luke Verstraten [b], Philippa Bryant [b], Bernhard J. Wehr [c], Neil Tindale [b], Neal W. Menzies [c], Ram C. Dalal [c,d]

[a] Dept of Agriculture and Fisheries, Queensland Government, University of the Sunshine Coast, Sippy Downs, 4556, Australia
[b] Faculty of Science, Health, Education and Engineering, University of the Sunshine Coast, Sippy Downs, 4556, Australia
[c] School of Agriculture and Food Sciences, The University of Queensland, St Lucia 4072, Australia
[d] Dept of Science, Information Technology and Innovation (DSITI), Queensland Government, Ecosciences Precinct, 41 Boggo Rd, Dutton Park, 4102, Australia

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ABSTRACT

Land-use change can have a major influence on soil organic carbon (SOC) and above-ground C pools. We assessed a change from native vegetation to introduced *Pinus* species plantations on C pools using eight paired sites. At each site we determined the impacts on 0–50 cm below-ground (SOC, charcoal C, organic matter C, particulate organic C, humic organic C, resistant organic C) and above-ground (litter, coarse woody debris, standing trees and woody understorey plants) C pools. In an analysis across the different study sites there was no significant difference ($P > 0.05$) in SOC or above-ground tree C stocks between paired native vegetation and pine plantations, although significant differences did exist at specific sites. SOC (calculated based on an equivalent soil mass basis) was higher in the pine plantations at two sites, higher in the native vegetation at two sites and did not differ for the other four sites. The site to site variation in SOC across the landscape was far greater than the variation observed with a change from native vegetation to introduced *Pinus* plantation. Differences between sites were not explained by soil type, although tree basal area was positively correlated with 0–50 cm SOC. In fact, in the native vegetation there was a significant linear relationship between above-ground biomass and SOC that explained 88.8% of the variation in the data. Fine litter C (0–25 mm diameter) tended to be higher in the pine forest than in the adjacent native vegetation and was significantly higher in the pine forest at five of the eight paired sites. Total litter C (0–100 mm diameter) increased significantly with plantation age ($R^2 = 0.64$). Carbon stored in understorey woody plants (2.5–10 cm DBH) was higher in the native vegetation than in the adjacent pine forest. Total site C varied greatly across the study area from 58.8 Mg ha^{-1} at a native heathland site to 497.8 Mg ha^{-1} at a native eucalypt forest site. Our findings suggest that the effects of change from native vegetation to introduced *Pinus* sp. forest are highly site-specific and may be positive, negative, or have no influence on various C pools, depending on local site characteristics (e.g. plantation age and type of native vegetation).

1. Introduction

There is global concern that land-use change results in a depletion of soil organic carbon (SOC), terrestrial biomass and consequent increases in atmospheric CO_2 (e.g. Houghton, 2003; Strassmann et al., 2008). Conversion of forest to agricultural land-uses usually results in loss of above-ground biomass C and SOC, particularly when conversion is to cultivated land (Brown and Lugo, 1990; Ellert and Gregorich, 1996; Murty et al., 2002; Guo and Gifford, 2002). However, the impacts of change from one forest type to another are less clear and there is a high degree of uncertainty regarding the degree and direction of change (Bashkin and Binkley, 1998; Rhoades et al., 2000) due to factors such as plantation age, type of plantation (native or exotic species), soil type and environmental factors (e.g. climate) and management factors (Kasel and Bennett, 2007).

The role of different forest compositions on forest C stocks and dynamics is poorly understood (Jandl et al., 2007) and there is a paucity of detailed information on soil C stocks in sub-tropical forests. An international review by Guo and Gifford (2002) reported

* Corresponding author at: Dept of Agriculture and Fisheries, Queensland Government, University of the Sunshine Coast, Sippy Downs, 4556, Australia.
E-mail address: tom.lewis@daf.qld.gov.au (T. Lewis).

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that, on average, SOC stocks declined by 15% following conversion from native forest to conifer plantation, but there was variability in this change depending on plantation species, rainfall and plantation age. Studies in southern Australia (e.g. Turner and Lambert, 2000; Turner et al., 2005) also reported reductions in SOC stocks following land-use conversion to conifer plantations. Further, in the sub-tropics, Chen et al. (2004) and Richards et al. (2007) reported significant reductions in total SOC following conversion of native vegetation to hoop pine (*Araucaria cunninghamii*, a native species to the region) plantations. The losses in SOC are likely due, at least in part, to site preparation for tree planting, which involves cultivation (e.g. ripping and mounding) that disturbs soil structure and breaks down soil aggregates (Jandl et al., 2007). However, few published studies have reported the impacts of conversion to introduced conifer plantations in the sub-tropics, and few studies have considered the form of soil organic C (i.e. humic, particulate or resistant) which is important when considering the resilience of these C stocks and our ability to model SOC changes.

Following clearing of native forest there is an initial reduction in above-ground plant biomass. In the case where native forest is replaced with plantation forest, above-ground biomass may be lower (Chen et al., 2005), similar, or reach higher levels than in the previous vegetation (e.g. Lugo, 1992), usually through modification of the site productivity (e.g. addition of fertilizer, Oren et al., 2001). Reports of the amount of above-ground biomass for *Pinus* plantations in the sub-tropics suggest approximately 316 Mg ha^{-1} (~155 Mg C ha^{-1}) can be sequestered by age 30 (Simpson et al., 2000). Tree C stocks in native vegetation in sub-tropical Australia may vary greatly from site-to-site but range from approximately 40 to 220 Mg C ha^{-1} depending on the soil type and site productivity (Westman and Rogers, 1977; Hero et al., 2013; Ngugi et al., 2014; Moroni and Lewis, 2015). We are unaware of any studies that consider the impact of conversion of native vegetation to *Pinus* sp. plantation on tree C and litter C stocks in the sub-tropics of Australia.

In both native vegetation and plantation forest, a significant C stock can be found in the litter layer, which potentially plays an important role in building soil C (Liski et al., 2002). This C pool may be dynamic (Bubb et al., 1998; Birk and Simpson, 1980), but often a steady state between litterfall and decomposition is reached over time (Olson, 1963) and incorporating litter C into total site C stocks can be important when assessing land-use changes (Richter et al., 1999; Paul et al., 2002). *Pinus* sp. forests are known to contain particularly high litter biomass stocks, partly due to slower rates of decomposition relative to native forests (e.g. Paul and Polglase, 2004; Prescott, 2010), however, there are few published comparisons between *Pinus* sp. forests and native vegetation in tropical and sub-tropical regions.

The commercial plantation forestry estate covers approximately two million hectares in Australia, of which 51% is planted with softwood species (Montreal Process Implementation Group for Australia and National Forest Inventory Steering Committee, 2013) with approximately 18% of this softwood estate occurring in Queensland. This study focusses on the impacts of change from native vegetation to plantation forest, which occurred 28–60 years ago, using paired comparison sites in Queensland. We aimed to determine, across a range of sites with varying soil types and plantation ages, whether C pools differed between introduced *Pinus* plantations and adjacent native vegetation. Based on the above-mentioned studies in southern Australia we hypothesised that SOC would be lower in the conifer plantations, but litter C would be higher in these plantations relative to the native vegetation. We also hypothesised that above-ground woody plant C would be lower in the *Pinus* plantations, particularly in young plantations, given the likely relationship between above-ground biomass and plantation age.

2. Methods

2.1. Site details

Eight paired sites were selected from within the *Pinus* sp. plantation resource in south-eastern Queensland, Australia (Fig. 1). *Pinus* sp. plantations in the paired comparisons varied from six to 34 years since planting (mean age of 21 years, Table 1). Plantation plots were either in their first or second rotation (Table 1). Soil types varied between the eight sites (classified using Australian Soil Classification, Isbell, 1996): two sites were on yellow Kandosols, two sites were on brown Kandosols, one site was on a grey Chromosol and three sites were on Podosols (Table 1). Pine plantations were dominated by *Pinus elliottii* var. *elliottii*, *Pinus caribaea* var. *hondurensis* and *P. elliottii* × *P. caribaea* hybrids (Table 1). Adjacent native vegetation varied between sites; in most cases (six sites) it was naturally occurring forest dominated by tree species including *Eucalyptus racemosa*, *Corymbia intermedia*, *Eucalyptus acmenoides*, *Lophostemon suaveolens*, *Syncarpia glomulifera* and *Melaleuca quinquenervia*, or open woodland (one site) and heathland (one site) with dominant species such as *Eucalyptus umbra*, *Banksia aemula* and *Melaleuca viridiflora* (Table 1). In all cases the native vegetation was multi-aged remnant vegetation and hence could not be accurately aged. Mean annual rainfall across the study region varied from 1193 to 1611 mm (average = 1408 mm, Table 1), with rainfall being higher in the summer months. Mean minimum temperature ranged from 14.7 °C to 15.8 °C while mean maximum temperature ranged from 25.1 °C to 26.6 °C. Climatic data for the study area were based on spatially interpolated Bureau of Meteorology observational data from 1889 to 2013 (Jeffrey et al., 2001).

2.2. Plot layout

Paired-comparison sites were chosen on the basis of sufficient area of the target vegetation being on the same soil type, with the same slope position. Plots were 0.5 ha (in most cases 100 × 50 m) in both the pine and native vegetation, and were separated by <200 m at each site. Six of the eight paired sites were selected as part of an earlier (unpublished) study in 1998 and the authors are currently investigating temporal trends in soil nutrients over this time. Each plot was divided into 50 sub-plots of 10 × 10 m and six sub-plots were randomly selected for sampling (stratified simple random sampling, Fig. 1b). Plots and sub-plots were established using tape measures, optical squares and sighting posts to ensure right-angles. Each sub-plot contained 100 1 × 1 m squares, of which ten were randomly selected for sampling (e.g. Fig. 1c). Each selected sub-plot and square was marked with line-marking paint to delineate the sampling positions. The positions of sub-plots and sampling squares were referenced from the plot corner positions to determine their UTM reference points and to allow future sampling within the same locations. Sampling took place between May and November 2013.

2.3. Litter sampling and above-ground carbon estimates

A steel quadrat (0.5 × 0.5 m square) was placed in the centre of each 1 × 1 m sample square, and all dead and detached vegetation (litter) was collected down to the soil surface, being careful to exclude mineral soil. All litter material ⩽25 mm diameter was defined as fine litter and litter material >25 mm and <100 mm diameter was defined as coarse litter. All material collected within each sub-plot was bulked by litter type (i.e. fine and coarse) and fine litter was weighed in the field. A representative sub-sample (>25% of the total biomass) of the fine litter from each sub-plot was placed in a paper bag, weighed in the field

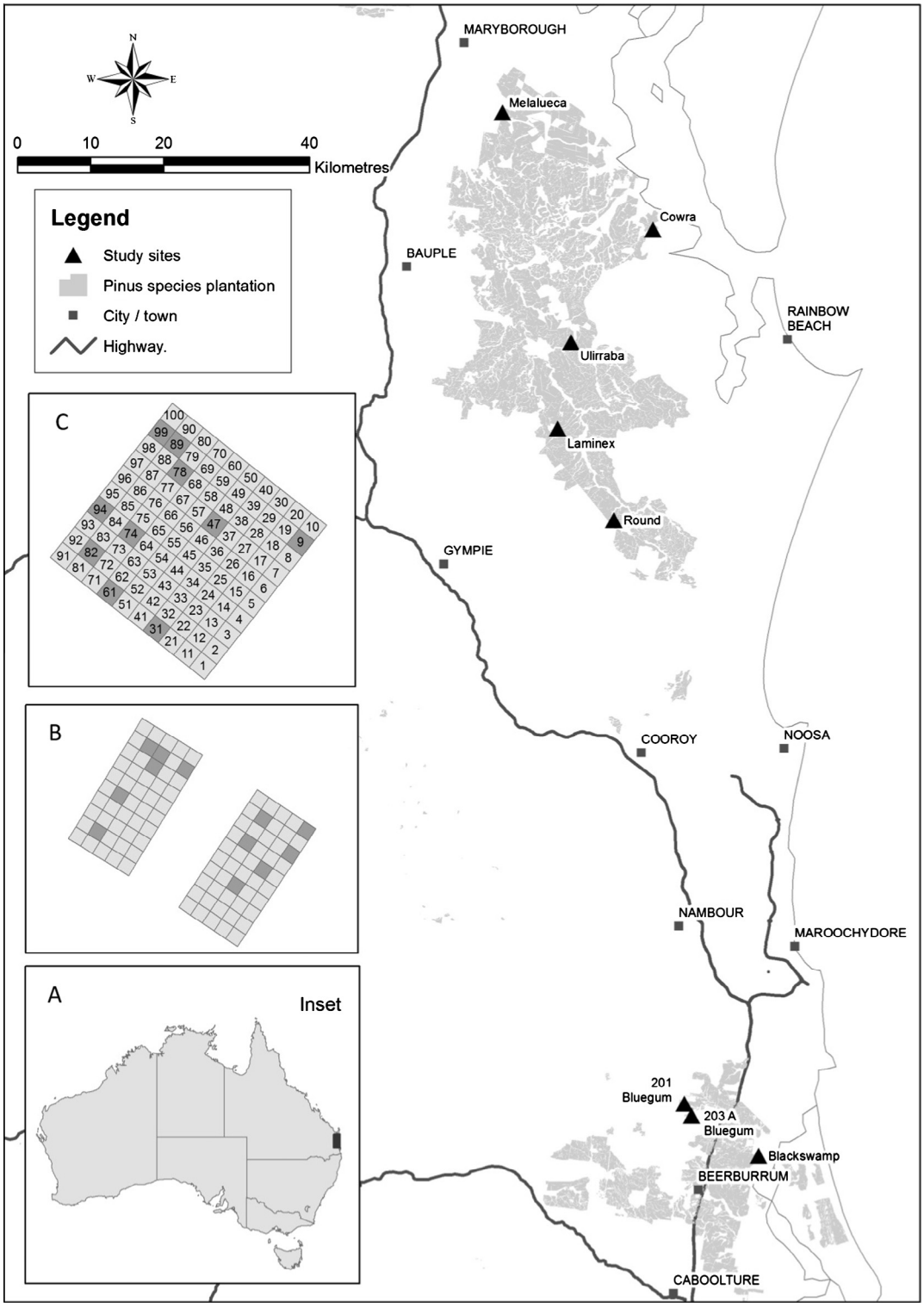

Fig. 1. Locations of paired sampling sites (pine forest and adjacent native vegetation) within the pine plantation estate of southern Queensland, Australia (inset A). Examples are shown of: the plot sampling layout (inset B), where darker shaded cells represent those randomly selected for sampling; and inset C, showing locations of 10 sampling points randomised within a 10 × 10 m sub-plot.

then oven dried at 65 °C to constant weight. This provided a moisture correction factor for total field weights of fine litter. All coarse litter was collected and oven dried (65 °C) to constant weight. Litter C was determined by multiplying biomass by C concentration. Litter C concentration was determined by dry-combustion with a LECO CNS-2000 analyser (LECO Corporation, MI, USA).

Material ⩾100 mm was defined as coarse woody debris (CWD) and was assessed using the line-intersect method (Van Wagner, 1968; McKenzie et al., 2000). This involved running a series of 10 m transects from a random start point and initial transect direction. At each 10 m interval, the transect continued at 90° to the current direction of travel, where direction of travel (left or right) was determined randomly. If the transect inter-

Table 1
Details of the study sites in south-eastern Queensland Australia. Rotation refers to whether the plantation is in its first or second *Pinus* sp. rotation following conversion from native vegetation. Plantation age is based on the number of years since planting in the current rotation. Soil type is based on Isbell (1996). Tree basal area (BA) was calculated for all live trees (DBH trees ⩾ 10 cm) within a 0.5 ha plot. Mean annual rainfall (MAR, mm) is provided for each site.

Site name	Dominant species	Plantation age (years)	Rotation	Soil type	Tree BA (m^2 ha^{-1})	MAR (mm)
Blackswamp	*Pinus elliottii* var. *elliottii*	34.1	1st	Podosol	43.4	1495
	Eucalyptus racemosa, *Corymbia intermedia*, *Melaleuca quinquenervia*	na	na	Podosol	30.2	
Bluegum 201	*Pinus caribaea* var. *hondurensis*	28.5	2nd	Brown Kandosol	41.3	1611
	Syncarpia glomulifera, *C. intermedia*, *Lophostemon suaveolens*, *Eucalyptus cloeziana*	na	na	Brown Kandosol	60.2	
Bluegum 203	*P. elliottii* × *P. caribaea*	26.0	2nd	Yellow Kandosol	54.6	1611
	E. racemosa, *Eucalyptus acmenoides*, *C. intermedia*, *L. suaveolens*	na	na	Yellow Kandosol	22.1	
Cowra	*P. caribaea* var. *hondurensis*	26.5	1st	Podosol	41.7	1358
	Eucalyptus umbra, *Banksia aemula*	na	na	Podosol	10.0	
Laminex	*P. elliottii* × *P. caribaea*	9.7	2nd	Brown Kandosol	31.9	1228
	E. racemosa, *E. acmenoides*, *C. intermedia*	na	na	Brown Kandosol	22.7	
Melaleuca	*P. elliottii* × *P. caribaea*	6.4	2nd	Grey Chromosol	5.2	1193
	Eucalyptus umbra, *Melaleuca viridflora*	na	na	Grey Chromosol	3.4	
Round	*Pinus elliottii* var. *elliottii*	31.5	1st	Podosol	47.5	1520
	Eucalyptus tereticornis, *C. intermedia*, *Melaleuca quinquenervia*, *L. suaveolens*	na	na	Podosol	37.0	
Ulirraba	*P. elliottii* × *P. caribaea*	8.3	2nd	Yellow Kandosol	30.9	1251
	E. acmenoides, *C. intermedia*, *S. glomulifera*, *E. racemosa*	na	na	Yellow Kandosol	13.2	

sected with the plot edge, then the direction of travel rebounded 90° at that point back into the plot for the remaining part of the 10 m transect (McKenzie et al., 2000). Each transect was continued for a total distance of 50 m and three 50 m transects were conducted in each plot. Diameters were recorded for any CWD that intersected the transect and an estimation of decay (%) recorded for each piece.

CWD volume V was calculated using:

$$\text{Volume} = (\pi^2 \Sigma\, d^2/8L)$$

where V is volume of wood per unit area (m^3 m^{-2}), d is piece diameter (m) and L is length of transect line (m) (Van Wagner, 1968). This formula assumes that pieces were cylindrical, horizontal and randomly oriented. For CWD with decay, volume calculations involved subtraction of the decayed proportion of each piece (i.e. missing volume in reference to a simple cylindrical volume). To convert CWD volumes to a mass per unit area, the volumes of intact and decayed CWD were multiplied by their respective wood densities (450 kg m^{-3} for pines and 650 kg m^{-3} for native vegetation, Ilic et al., 2000). Carbon concentration of CWD was not measured, but C stocks of CWD were estimated by multiplying CWD mass (Mg ha^{-1}) by a default C concentration value of 50% (Coomes et al., 2002; Garrett et al., 2008).

For each plot the diameter at breast height (DBH) of trees ⩾10 cm was measured. The DBH was used to calculate the basal area (m^2 ha^{-1}) of woody vegetation and to provide an estimation of the above-ground biomass. The above-ground biomass was estimated using general allometric relationships for softwood plantations and eucalypt vegetation for the *Pinus* sp. forest and adjacent vegetation, respectively (Paul et al., 2016). These allometrics were developed based on existing biomass datasets in Australia and were based on 455 individuals of softwoods (mostly *Pinus radiata*) and 6004 eucalypt individuals (*Eucalyptus* and closely-related genus of *Corymbia* and *Angophora*). For the *Pinus* sp. plantations and native vegetation we used the following equations, respectively:

(1) Above-ground biomass (kg) = exp [−2.573 + 2.460 ln (DBH)] × 1.018
(2) Above-ground biomass (kg) = exp [−2.016 + 2.375 ln (DBH)] × 1.067

Tree biomass was converted to carbon using a C concentration of 49% (Gifford, 2000). In each sub-plot the diameter of all woody plants with a DBH 2.5–9.9 cm were measured to allow calculation of sub-plot basal area and to estimate C stored in these understorey plants. No assessment of biomass was made for grasses, herbs and vines. Biomass of these components is usually <3 Mg ha^{-1} in the ecosystems studied (Moroni and Lewis, 2015; Westman and Rogers, 1977).

2.4. Soil sampling

Following litter collection, at each randomly selected sampling location soil samples were collected to a depth of 50 cm using 70 cm long hardened steel cores with two different cutting head sizes. Cutting heads were 42 mm and 44 mm, with all cores having an internal tube diameter of 45 mm. For fine textured soils, the 42 mm cutting heads were used due to expansion of clays within the tube, and for coarse textured sandy soils the 44 mm cutting heads were used, as expansion of sands within the tube is usually minimal. The cores were driven into the ground using a Bosch GSH16 jack-hammer powered by a portable generator (Honda EU20i 240 V). A specially designed soil-core lifter was used to remove the core from the ground.

The soil samples were pushed out of the core onto hemi-cylindrical tubes, then divided into five sampling depths: 0–5 cm, 5–10 cm, 10–20 cm, 20–30 cm and 30–50 cm and transferred into labelled, sealable plastic bags. Soil samples collected within each of the ten 10 × 10 m sub-plot were bulked together for each depth. Once collected, soils were kept in a cool dark location until the samples were air dried, processed and sent to the laboratory.

In addition to the soil C samples, samples for 'soil core mass' (oven-dried mass per unit core volume for bulk density) determinations were collected from four randomly selected, previously sampled squares in each plot. Each of these samples was collected using the same core sampler as that used for soil C samples. Soil core mass samples were collected for the same sampling depths as for the standard soil samples, and were placed in individually labelled plastic bags for each depth. These samples were later dried in an oven at 40 °C to constant weight, to determine air-dry weights, and then dried at 110 °C to constant weight, to determine the oven-dry weight for calculation of core mass and the moisture correction factor between air-dry and oven-dry soil for a plot.

2.5. Soil processing and analysis

All soil samples, except the core mass samples, were weighed after air drying and carefully processed by hand through a 2 mm sieve. During processing, visible organic material (roots, buried debris, fungal hyphae and macro fauna including material <2 mm), charcoal and rocks were separated by hand using long nosed surgical tweezers. While we did separate visible plant roots, it is acknowledged that our sampling did not attempt to sample the 'root ball' and tap roots located immediately below the tree bole and it is unlikely that our sampling intensity (60 cores per 0.5 ha) was adequate to accurately sample total root biomass (Resh et al., 2003). The contribution of this root mass is known to be variable, but significant (e.g. Westman and Rogers, 1977; Resh et al., 2003; Coll et al., 2008) and hence our values of below-ground C are under-estimates of total below-ground C. We did not attempt to sample this below-ground 'root ball' biomass pool due to the destructive nature of such sampling, nor did we estimate this pool as there is little available root biomass data for sub-tropical *Pinus* sp. forests and adjacent native vegetation.

The components removed during processing were oven-dried at 65 °C for 48 h and weighed. The oven-dried weights of these components were subtracted from the moisture corrected total sample weight, to give the oven-dried weight of the <2 mm fraction of the soil sample. The soil density was calculated for each soil sample using the oven-dried weight of the <2 mm soil sample and the volume of the soil coring tube at the individual samples depth interval. The soil sample was also inspected for the presence of carbonates, but no carbonates were detected in the soils in this study (soil pH 4.5–6). Sub-samples of the <2 mm soil were separated from the whole sample by passing repeatedly through a riffle box sample splitter. Total C and nitrogen (N) concentrations were then determined by dry-combustion with a LECO CNS-2000 analyser (LECO Corporation, MI, USA). Soil organic C fractions for particulate (POC), humus (HOC), and resistant (ROC) were estimated using mid-infrared spectroscopy (MIRS) following the methods of Baldock et al. (2013).

SOC stocks for each depth interval were calculated using:

$$\text{SOC (Mg ha}^{-1}) = \%\text{C} \times \rho \times V \times (1 - f)$$

where %C is the C concentration (% weight); ρ the soil density (g m^{-3}); and V the volume (m^3) of soil per hectare (depth in m $\times$ 10^4 m^{-2}) in the samples depth interval, after the volume fraction (f) of the organic material, charcoal and rocks have been subtracted.

2.6. Data analysis

Analysis of variance (ANOVA) was carried out in GenStat (16th edition, VSN International Ltd.) to determine the effects of vegetation type (pine plantation *vs* native vegetation) on soil C stocks across the eight sites. As the five different depth levels could not be randomised, and there is likely some correlation between values down the profile, we treated the depth factor as a 'repeated measures' factor, which takes into account levels of correlation and makes appropriate adjustments to tests of significance and least significant difference (LSD) values. Soil response variables analysed included SOC (Mg ha^{-1}), charcoal C (Mg ha^{-1}), visible organic material C (Mg ha^{-1}), below-ground C (SOC plus charcoal C plus visible organic material C, Mg ha^{-1}), MIR POC (Mg ha^{-1}), MIR HOC (Mg ha^{-1}) and MIR ROC (Mg ha^{-1}). These variables were converted to a comparable depth (5 cm) prior to analysis to account for the different volumes of soil in the different depth intervals. Analysis of the soil variables was carried out on a fixed depth interval basis and on an equivalent soil mass basis (Ellert and Bettany, 1995) due to likely modifications in density associated with plantation establishment. Equivalent soil mass for each depth interval was calculated using the cubic spline method of Wendt and Hauser (2013). In addition to the above soil variables, we analysed litter C (fine and coarse, Mg ha^{-1}), CWD C (Mg ha^{-1}), small woody plant C (Mg ha^{-1}), plot above-ground tree C (Mg ha^{-1}) and total ecosystem C (0–50 cm below-ground C plus above-ground woody plant C plus all litter and CWD, Mg ha^{-1}) using a split-plot ANOVA (with variation due to sites accounted for in the 'Block' stratum). In addition to analysing soil variables at individual depths, variables were also analysed for the total 0–30 cm and 0–50 cm depths using a split-plot ANOVA, with soil type included as a factor.

To account for variations in plantation age, tree basal area (all plants with a DBH > 2.5 cm) was included as a covariate in the analyses, where significant. Where necessary, variables were log transformed ($\log_{10} + 1$) to meet the assumptions of ANOVA. ANOVA was also used to determine significant vegetation type effects and vegetation type $\times$ depth effects at an individual site level. Regression analysis was used to investigate relationships between response variables and potential predictors (e.g. plantation age, tree basal area, mean annual rainfall). Predicted means and LSDs (5% level) from ANOVA analyses and adjusted R^2 from regression analysis are reported.

3. Results

3.1. Soil and below-ground C pools

Soil charcoal C stocks were significantly higher in the pine forest at three of the eight sites and significantly higher in the native vegetation at two of the eight sites while the remaining three sites did not differ significantly in charcoal stocks (Table 2). Across all sites, soil charcoal C stocks did not differ between the native and non-native vegetation types ($F_{1,7} = 1.04$, $P = 0.34$, Table 3), the vegetation type $\times$ depth interaction was not significant ($F_{4,56} = 0.34$, $P = 0.75$) and tree basal area had no significant influence. Soil charcoal C stocks did vary with depth ($F_{4,56} = 15.71$, $P < 0.001$); with greater stocks in the upper soil layers (predicted means of 0.25, 0.26, 0.15, 0.07 and 0.03 log + 1 Mg ha^{-1} per 5 cm depth, for the 0–5, 5–10, 10–20, 20–30 and 30–50 cm depths, respectively, LSD = 0.13). Mean charcoal C stocks for 0–30 cm depth were 2.9 Mg ha^{-1} for the native vegetation and 4.6 Mg ha^{-1} for the pine forest (LSD = 2.7).

Visible organic matter was significantly higher in the native vegetation than in the pine forest at two of the eight sites, was significantly higher in the pine forest at one site and did not differ significantly at the remaining sites (Table 2). Across all sites, only depth of sample had a significant influence on visible organic matter C stocks ($F_{4,56} = 49.1$, $P < 0.001$). Visible organic matter C stocks were higher at the surface depths (predicted means of 0.69, 0.60, 0.50, 0.37, 0.22 log + 1 Mg ha^{-1} per 5 cm depth, for the 0–5, 5–10, 10–20, 20–30 and 30–50 cm depths, respectively, LSD = 0.10). Across all sites, tree basal area had no influence on the visible organic matter pool, but had a significantly positive association at two individual sites (Table 2). Mean visible organic matter carbon stocks for 0–30 cm depth were 16.0 Mg ha^{-1} for the native vegetation and 15.2 Mg ha^{-1} for the pine forest (LSD = 8.8).

SOC (equivalent mass) was significantly higher in the pine forest at two of the eight sites, was significantly higher in the native vegetation at two sites and did not differ between native and non-native vegetation types at four sites (Table 2). Across all sites, SOC stocks did not differ between native and non-native vegetation types ($F_{1,6} = 0.48$, $P = 0.51$, Table 3) and the vegetation type - $\times$ depth interaction was not significant ($F_{4,56} = 0.47$, $P = 0.64$). However, SOC stocks did differ among depths ($F_{4,56} = 23.0$,

Table 2
Summary of ANOVA results for individual paired sites (*Pinus* forest, PF *vs* native vegetation, NV) for below-ground C pools (0–50 cm). Sites at which significant differences ($P < 0.05$) occurred for the main effects of vegetation type (pine and native vegetation) are identified with an $*$. Sites at which significant differences occurred for depth (0–5, 5–10, 10–20, 20–30 and 30–50 cm) and the vegetation type by depth interaction are listed. Predicted means (Mg ha^{-1}) are presented and where analyses where run on log transformed data means were back-transformed. 'Site' refers to site code (BSW, Blackswamp; BG03, Bluegum 203; BG01, Bluegum 201; ULI, Ulirraba; LAM, Laminex; COW, Cowra; MEL, Melaleuca; RND, Round); 'Trend' refers to the overall trend with depth from the surface to 50 cm. The number of individual sites with significant positive or negative associations with tree basal area are listed under the 'Covariate' column.

Carbon pool	Vegetation type			Depth		Significant Vegetation type × Depth interactions			Covariate (tree BA)
	Site	NV mean	PF mean	Sites at which differences exists	Trend	Site and depths at which differences exists	NV mean	PF mean	
Soil organic C (ESM)	BSW	7.5	8.6	BSW	All ↓	BG01, 0–5 cm	4.2	11.3	1 +ve (BSW)
	BG03*	6.3	5.0	BG03		30–50 cm	10.3	4.3	
	BG01	8.7	7.7	ULI		ULI, 0–20 cm	15.5	26.6	
	ULI	2.8	4.1	LAMCOW		COW, 0–10 cm	15.3	7.6	
	LAM*	5.5	5.2	MEL		30–50 cm	1.2	2.8	
	COW	4.8	3.2	RND		MEL, 0–10 cm	4.8	6.4	
	MEL*	1.3	1.5						
	RND	5.8	6.1						
Humic organic C	BSW*	2.5	3.9	BG03	All ↓	ULI, 20–50 cm	1.9	1.1	1 −ve (RND)
	BG03	2.7	2.6	BG01					
	BG01*	1.8	1.4	ULI					
	ULI*	1.7	1.5	LAM					
	LAM	3.1	3.2	COW					
	COW*	0.8	0.9	MEL					
	MEL	1.5	1.6	RND					
	RND	3.6	3.9						
Particulate organic C	BSW	1.3	1.3	All sites	All ↓	ULI, 10–50 cm	1.1	2.2	2 +ve (BSW, BG03)
	BG03*	1.0	0.7			COW, 0–20 cm	4.1	0.7	
	BG01	1.1	1.3			MEL, 0–10 cm	0.6	1.4	
	ULI*	0.8	1.0			RND, 0–20 cm	5.7	4.3	
	LAM	1.0	0.8						
	COW*	0.9	0.2						
	MEL*	0.2	0.4						
	RND*	1.4	1.1						
Resistant organic C	BSW	2.7	3.4	BSW	All ↓	MEL, 0–20 cm	0.4	1.2	1 +ve (BSW)
	BG03*	1.4	1.2	BG03					1 −ve (RND)
	BG01	2.5	2.2	BG01					
	ULI*	1.2	1.7	ULI					
	LAM	1.2	1.1	LAM					
	COW	0.3	0.5	MEL					
	MEL*	0.1	0.3	RND					
	RND	1.3	1.7						
Charcoal C	BSW*	0.78	0.37	BG03	All ↓				1 +ve (BG03)
	BG03*	0.44	0.37	BG01					1 −ve (COW)
	BG01	0.98	0.75	ULI					
	ULI*	0.58	1.05	LAM					
	LAM	0.28	0.27	COW					
	COW*	0.00	0.48	MEL					
	MEL	0.05	0.03	RND					
	RND*	0.02	0.07						
Visible organic matter C	BSW*	1.3	1.6	All sites	All ↓	BSW, 0–10 cm	4.4	8.0	3 +ve (LAM, MEL, RND)
	BG03*	1.2	1.3			ULI, 0–10 cm	2.9	9.5	
	BG01	0.8	1.0			30–50 cm	0.3	0.1	
	ULI*	0.9	1.5			BG01, 0–10 cm	2.5	4.4	
	LAM	2.6	2.4			COW, 0–20 cm	24.8	11.2	
	COW*	3.8	2.2			RND, 0–5 cm	1.4	3.2	
	MEL*	1.9	1.4			20–30 cm	1.2	0.8	
	RND	1.3	1.3						
Total below-ground C	BSW*	5.1	6.0	All sites	All ↓	ULI, 0–30 cm	25.9	46.7	2 +ve (BSW, MEL)
	BG03	9.1	8.5			BG01, 0–5 cm	8.7	17.2	
	BG01*	5.5	5.4			20–50 cm	6.3	4.0	
	ULI*	3.9	5.9			COW, 0–20 cm	41.4	22.5	
	LAM	10.1	9.3			RND, 0–5 cm	12.3	16.1	
	COW*	5.7	4.2						
	MEL*	2.9	2.7						
	RND*	4.6	4.9						

$P < 0.001$). SOC stocks decreased with depth; predicted means were 8.27, 7.45, 5.90, 4.20, 2.89 Mg ha^{-1} per 5 cm depth, for the 0–5, 5–10, 10–20, 20–30 and 30–50 cm depths, respectively (LSD = 1.49). Mean SOC stocks for 0–30 cm depth were 33.7 Mg ha^{-1} for the native vegetation and 36.7 Mg ha^{-1} for the pine forest (LSD = 10.6). Tree basal area was positively associated with SOC at one site (Table 2) and had an overall positive association with 0–50 cm SOC stock ($F_{1,6} = 25.83$, $P = 0.002$, $\beta = 5.09$). However, based on the limited number of data points for the plantation sites (eight points, ages 6–34 years), the relationship between SOC and current rotation age was not significant ($F_{1,7} = 2.60$, $P = 0.16$).

Below-ground C was significantly higher in the pine forest at three of eight sites, significantly higher in the native vegetation at three of the eight sites and did not differ between native and non-native vegetation types at two sites (Table 2). Across all sites,

Table 3
Summary of observed mean ± standard error (range) 0–50 cm below-ground C pools and above-ground C pools (Mg ha^{-1}) for pine plantations and adjacent native vegetation. Differences based on ANOVA analyses (sometimes on log transformed data) across all sites were non-significant in all cases except for small woody plant C ($P = 0.006$) and fine litter C, which was only marginally significant ($P = 0.09$).

Carbon pool (Mg ha^{-1})	Pine plantation	Native vegetation
Soil organic C (ESM)	55.9 ± 8.2 (17.4–86.8)	57.0 ± 12.2 (14.5–132.3)
Humic organic C	23.0 ± 3.5 (13.2–38.4)	21.7 ± 2.5 (9.5–32.2)
Particulate organic C	7.7 ± 1.2 (3.1–12.9)	9.4 ± 1.9 (1.5–18.2)
Resistant organic C	13.4 ± 2.8 (1.8–27.5)	11.6 ± 2.6 (0.7–20.8)
Charcoal C	4.3 ± 1.5 (0.2–13.4)	3.1 ± 1.1 (0.1–8.7)
Visible organic matter C	17.4 ± 1.7 (10.7–27.7)	19.7 ± 5.4 (9.5–55.8)
Total below-ground C	77.6 ± 8.4 (33.9–107.9)	79.8 ± 12.5 (33.3–153.2)
Fine litter C	15.5 ± 2.8 (5.5–30.6)	10.7 ± 2.0 (3.1–21.0)
Coarse litter C	1.9 ± 0.4 (0.6–4.0)	1.0 ± 0.3 (0.0–2.4)
Coarse woody debris C	3.2 ± 1.3 (0.3–9.4)	5.4 ± 2.0 (0.0–18.1)
Small woody plant C	0.5 ± 0.4 (0.0–3.1)	6.2 ± 1.5 (2.6–13.2)
Above-ground tree C	88.7 ± 14.3 (7.3–128.5)	83.7 ± 19.8 (2.3–183.0)
Total ecosystem C	243.3 ± 28.8 (77.6–344.8)	243.9 ± 45.1 (58.8–497.8)

below-ground C (SOC plus charcoal C plus visible organic material C) only differed significantly between depths ($F_{4,56} = 35.21$, $P = <0.001$). Predicted means were 14.1, 13.5, 10.0, 6.7, 4.4 Mg ha^{-1} per 5 cm depth, for the 0–5, 5–10, 10–20, 20–30 and 30–50 cm depths, respectively (LSD = 2.36). The effects of vegetation type ($F_{1,6} = 0.08$, $P = 0.79$, Table 3) and the vegetation type × depth interaction were not significant ($F_{4,56} = 0.51$, $P = 0.57$). Tree basal area was positively associated with the below-ground C pool ($F_{1,6} = 10.37$, $P = 0.018$, $\beta = 3.22$) across all sites, and at one of the eight individual sites (Table 2). Mean below-ground C stocks for 0–30 cm depth were 59.7 Mg ha^{-1} for the native vegetation and 62.5 Mg ha^{-1} for the pine forest (LSD = 21.0).

Humic organic carbon (HOC) was significantly higher in the pine forest at two of the eight sites, significantly higher also in the native vegetation at two of the eight sites and did not differ between native and non-native vegetation types at four sites (Table 2). Across all sites, the effect of vegetation type ($F_{1,7} = 0.25$, $P = 0.63$) on HOC and the vegetation type × depth interaction were again not significant ($F_{4,56} = 0.05$, $P = 0.95$). HOC did vary significantly between depths ($F_{4,56} = 25.40$, $P < 0.001$). Predicted means were 3.13, 3.12, 2.70, 2.10, 1.63 Mg ha^{-1} per 5 cm depth, for the 0–5, 5–10, 10–20, 20–30 and 30–50 cm depths, respectively (LSD = 0.42). Mean HOC stocks for 0–30 cm depth were 15.5 Mg ha^{-1} for the native vegetation and 16.2 Mg ha^{-1} for the pine forest (LSD = 3.4). Tree basal area was not associated with HOC across all sites and had a negative association with HOC at one individual site (Table 2).

Particulate organic carbon (POC) was significantly higher in the pine forest at two of eight sites, significantly higher in the native vegetation at three of the eight sites and did not differ between native and non-native vegetation types at three sites (Table 2). Across all sites, only sampling depth had an influence on POC ($F_{4,56} = 24.39$, $P < 0.001$). Predicted means were 2.04, 1.39, 0.88, 0.45 and 0.62 Mg ha^{-1} per 5 cm depth, for the 0–5, 5–10, 10–20, 20–30 and 30–50 cm depths, respectively (LSD = 0.44). The vegetation type effect ($F_{1,6} = 0.08$, $P = 0.79$, Table 3) and the vegetation type × depth interaction were again not significant ($F_{4,56} = 0.21$, $P = 0.76$). Mean POC stocks for 0–30 cm depth were 6.3 Mg ha^{-1} for the native vegetation and 5.9 Mg ha^{-1} for the pine forest (LSD = 1.6). Tree basal area was positively associated with POC across all sites ($F_{1,6} = 11.21$, $P = 0.015$, $\beta = 3.35$) and had a positive influence on POC at two individual sites (Table 2). POC varied significantly with soil type ($F_{3,4} = 7.13$, $P = 0.044$); the Podosol sites had higher POC values than the grey Chromosol site (predicted means of 11.7 and 2.3 Mg ha^{-1}, LSD = 5.91).

Resistant organic carbon (ROC) was significantly higher in the pine forest at two of eight sites, significantly higher in the native vegetation at one of the eight sites and did not differ between native and non-native vegetation types at five sites (Table 2). Across all sites, sampling depth had a significant influence on ROC stocks ($F_{4,56} = 28.99$, $P < 0.001$); predicted means were 2.39, 2.29, 1.76, 1.03, 0.57 Mg ha^{-1} per 5 cm depth, for the 0–5, 5–10, 10–20, 20–30 and 30–50 cm depths, respectively (LSD = 0.46). The vegetation type effect ($F_{1,6} = 1.14$, $P = 0.33$) and the vegetation type × depth interaction were again not significant ($F_{4,56} = 0.09$, $P = 0.94$). Mean ROC stocks for 0–30 cm depth were 9.7 Mg ha^{-1} for the native vegetation and 10.8 Mg ha^{-1} for the pine forest (LSD = 3.5). Tree basal area had a significant positive association with ROC across all sites ($F_{1,6} = 9.04$, $P = 0.024$, $\beta = 2.99$), had a positive association with ROC at one individual site, and a negative association at one site (Table 2).

3.2. Above-ground C pools

Fine litter C was significantly higher in the pine forest at five sites and higher in the native vegetation at one site (Table 3). Across all paired sites fine litter C varied marginally between the pine forest and native vegetation ($F_{1,7} = 3.92$, $P = 0.088$). Fine litter C tended to be higher in the pine forest than in the adjacent native vegetation (predicted means of 15.5 and 10.7 Mg ha^{-1}, LSD = 5.8, Table 3). Across all sites there was no significant difference in coarse litter C stocks between the pine forest and native vegetation ($F_{1,7} = 3.19$, $P = 0.12$, Table 3), although pine forest had greater coarse litter C than native vegetation at four individual sites (Table 3). Predicted mean coarse litter stocks across all sites were 1.0 Mg ha^{-1} in the native vegetation and 1.9 Mg ha^{-1} in the pine forests (standard error = 0.35 Mg ha^{-1}). Total litter C (fine plus coarse) increased significantly with plantation age in the pine forest ($F_{1,6} = 13.35$, $P = 0.011$, $R^2 = 0.64$; Fig. 2a) and with tree basal area across both vegetation types ($F_{1,14} = 8.48$, $P = 0.011$, $R^2 = 0.33$). The variation in litter C stocks among the native forest sites was almost as high as that among the pine plantation sites of varying age (Table 3), reflecting the variation in vegetation and soil types across the different native forest sites. There was no difference in coarse woody debris stocks between the pine forest and native vegetation ($F_{1,7} = 0.55$, $P = 0.48$). Mean coarse woody debris stocks were 5.4 Mg ha^{-1} in the native vegetation and 3.2 Mg ha^{-1} in the pine forests (standard error = 2.2 Mg ha^{-1}).

Pine trees nearing the end of the commercial rotation had sequestered up to 128 Mg C ha^{-1}. Carbon stored in pine plantation trees increased with plantation age (Fig. 2b, $F_{1,6} = 30.1$, $P = 0.002$) and plantation age explained 80.6% of the variation in above-ground tree C. The unexplained variation in this relationship is likely due to local site productivity factors, such as soil fertility and rainfall. Carbon stored in the trees ($\geqslant$10 cm DBH) in native vegetation adjacent to pine plantations varied from 2.3 Mg ha^{-1}

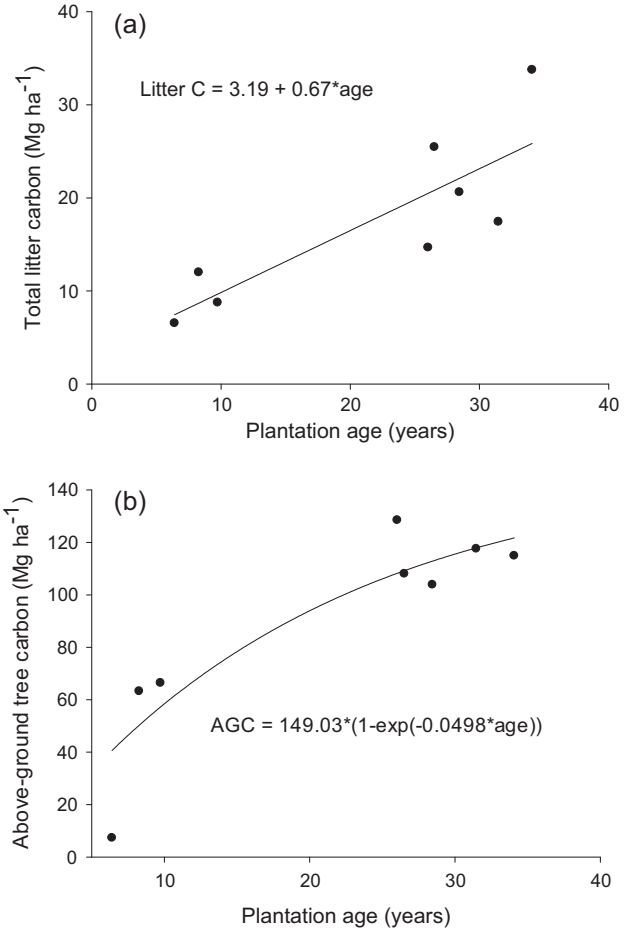

Fig. 2. Significant relationships between total litter C (fine and coarse litter) and pine plantation age (a, significant linear relationship, adjusted $R^2 = 0.64$) and between above-ground tree C and plantation age (b, significant exponential curve, adjusted $R^2 = 0.81$).

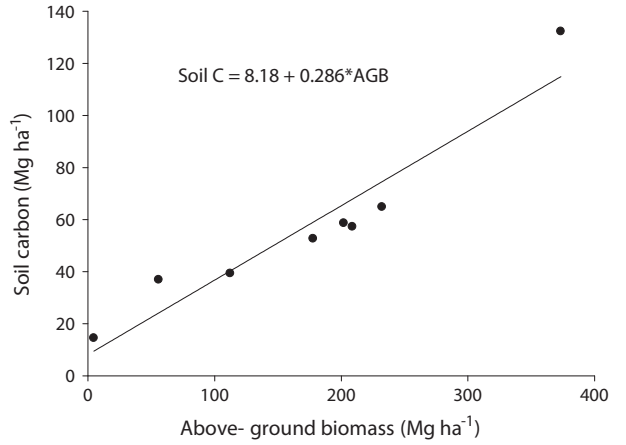

Fig. 3. Significant linear relationship between SOC and above-ground biomass (AGB) for the native vegetation plots that were paired with pine plantations (adjusted $R^2 = 0.89$).

in a heathland site to 183.0 Mg ha^{-1} in a eucalypt forest. Carbon stored in understorey woody plants (2.5–9.9 cm DBH) was a small pool, but was significantly higher in the native vegetation than in the pine forest ($F_{1,7} = 15.3$, $P = 0.006$, Table 3). There was a significant linear relationship between total 0–50 cm SOC and above-ground tree biomass for the native vegetation sites that explained 88.8% of the variation in the data ($F_{1,6} = 56.7$, $P < 0.001$, standard error of estimate = 11.5; Fig. 3). There was also a significant linear relationship between total 0–50 cm SOC in the native vegetation sites and mean annual rainfall ($F_{1,6} = 7.78$, $P = 0.032$, standard error of estimate = 16.2; $R^2 = 0.33$) which is to be expected given known the relationship between above-ground biomass and rainfall.

Above-ground C stocks varied greatly between individual sites (Fig. 4). Total ecosystem C (0–50 cm below-ground C plus above-ground woody plant C plus all litter and CWD) varied from 58.7 Mg ha^{-1} on the native heathland grey Chromosol site to 497.8 Mg ha^{-1} on a native eucalypt forest brown Kandosol site. Across all sites, there was no significant difference in total ecosystem C between the native vegetation and pine forests ($F_{1,7} = 0.00$, $P = 0.98$, Table 3).

4. Discussion

4.1. Below-ground C estimates

Our hypothesis, based on studies in southern Australia (Turner and Lambert, 2000; Turner et al., 2005) and meta-analyses (Guo and Gifford, 2002; Don et al., 2011), that SOC would be lower in *Pinus* plantations than adjacent native vegetation, was not supported across all sites. In fact, at two of eight sites SOC was higher in the plantation forests and this was driven by higher POC and ROC at both sites. Changes in POC are not surprising given these pools are known to respond to land-use changes (e.g. Chan, 2001; John et al., 2005). The differences in ROC between sites were unexpected as the resistant fractions are thought to be relatively inert and turn-over in longer time frames than those associated with vegetation type conversion in the current study (von Lützow et al., 2007). However, it is possible the ROC is higher in some pine plantations due to pyrogenic C, which may not be considered inert (Singh et al., 2012), as these plantations are frequently burnt with low intensity fire to reduce wildfire risk (Hunt and Simpson, 1985) and occurrence of fire to burn debris is common immediately following clearing of native vegetation. In fact at one site, both soil ROC and soil charcoal C that was separated during soil processing, was higher in the pine plantations than the adjacent native vegetation. At the two sites where SOC was greater in the plantation forest, the difference tended to be in the 0–20 cm depth. In fact, a further plantation site had higher SOC in the 0–5 cm depth horizon than in the native vegetation (i.e. significant vegetation type × depth interaction). This is contrary to the findings of a meta-analysis by Don et al. (2011) who reported higher SOC in the surface horizons of primary forest. This suggests that introduced *Pinus* plantations in the sub-tropics may have higher SOC and visible organic matter in the surface horizons than certain native vegetation communities, perhaps due to shallow and dense root systems (e.g. Mou et al., 1995) and associated mycorrhizae in these plantings.

Several studies report significant losses in SOC associated with forest type change from native forest to plantation forest (Guo and Gifford, 2002; Chen et al., 2004; Richards et al., 2007) although changes in SOC may be influenced by the species planted (Guo and Gifford, 2002; Don et al., 2011). In the current study, of the two sites where SOC was higher in the native vegetation relative to two 26 year old *Pinus* plantations, this difference was restricted to the 0–10 cm depths at one site, and was driven by higher POC. The native vegetation at this particular site was open woodland with dense clusters of proteoid roots near the soil surface (associated with *B. aemula*); which play an important role in the acquisition of phosphorus and other mineral nutrients in infertile soils (Dinkelaker et al., 1995; Lamont, 2003). At the other site where

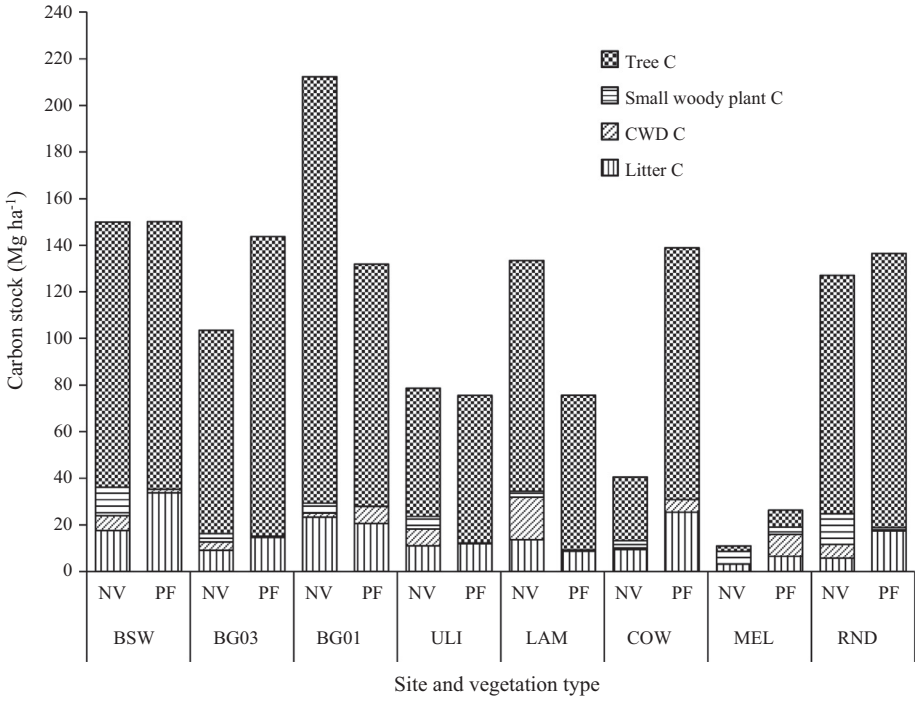

Fig. 4. Mean above-ground C pools for individual paired sites (NV, native vegetation and PF, pine forest), showing litter C (fine plus coarse litter), coarse woody debris C (CWD), small woody plant C (DBH 2.5–9.9 cm) and tree C (DBH ⩾ 10 cm). Site codes are: BSW, Blackswamp; BG03, Bluegum 203; BG01, Bluegum 201; ULI, Ulirraba; LAM, Laminex; COW, Cowra; MEL, Melaleuca; RND, Round.

SOC was higher in the native vegetation the difference was attributed to higher POC, ROC and charcoal C.

Reviews by Paul et al. (2002) and Guo and Gifford (2002) suggest it may take more than 30 or 40 years for SOC to be restored to its original levels following plantation establishment. Similarly, Turner and Lambert (2000) reported initial losses in SOC and suggested that at least 10–20 years is needed before net accumulation of SOC occurs after planting. Based on our chronosequence of different plantation ages, total SOC was not related to plantation age, and hence time since the most recent soil disturbance, although we acknowledge that additional data points or time-series data are needed to test this relationship properly. The findings reported here are corroborated by a concurrent study that re-visited six paired sites (native vegetation and *Pinus* sp. plantation) that were initially sampled in 1998 and then again 15 years later (unpublished data). Further, Gholz and Fisher (1982) reported relatively minor changes in SOC with increasing age in *P. elliottii* plantations in Florida. It appears that SOC varies greatly from site to site across the landscape, and that this variation is much greater than the variation observed with a change from native vegetation to introduced *Pinus* plantation. Thus changes in SOC associated with vegetation type change really need to be assessed at a site-level. The local site related factors that influence SOC stocks are currently not well understood although we speculate that the variable responses are at least partly due to variations in fine root biomass distribution and turnover in the different vegetation communities (Coleman et al., 2000; Rasse et al., 2005). At a landscape scale, certain environmental factors (e.g. rainfall, temperature, elevation, pH, soil texture, bulk density and vegetation type) and management factors (e.g. fertiliser additions) are known to be drivers of variation in SOC (Paul et al., 2002; Kasel and Bennett, 2007; Rabbi et al., 2014) and influence the site-to-site variation. The fact that our findings differ from some studies of *P. radiata* plantations in southern Australia might reflect the different *Pinus* species planted in the sub-tropics, the different environmental drivers (e.g. rainfall) and the high degree of variability between sites in our study (e.g. different plantation ages, site preparation and management methods, soil types, native vegetation types, etc.). Our findings are, however, supported by those of Kasel and Bennett (2007) who reported variable land-use change responses within different *P. radiata* plantations.

Our study was comprehensive in that it accounted for different below-ground C pools (organic material, charcoal, POC, HOC, ROC) in addition to SOC. However, separation into these pools had no major influence when testing for changes associated with vegetation type across all sites. Visible organic matter C, charcoal C and SOC all decreased with increasing depth down the profile, as expected based on previous studies (Gill et al., 1999; Jobbágy and Jackson, 2000; Guo and Gifford, 2002).

4.2. Above-ground and total ecosystem C estimates

Carbon stored in above-ground tree biomass was the largest C pool for most sites in this study (~35% of the total ecosystem C). Our estimates of tree biomass C are in the range of those reported for similar forest types in the region (Westman and Rogers, 1977; Simpson et al., 2000). Few studies have estimated total ecosystem C stocks, including SOC, for the study region. Our findings support those from elsewhere, that the greatest potential for C sequestration in plantation forests is through above- and below-ground tree biomass sequestration (Paul et al., 2002; Peichl and Arain, 2006). Our hypothesis that above-ground woody plant C would be lower in the *Pinus* plantations was only partially supported. At plantation sites that were nearing the end of the commercial rotation, tree C stocks were not always higher in the native vegetation (Fig. 4); in fact two such sites had higher tree C stocks in the plantation forest. Nevertheless, across all sites, C stored in trees was similar between the two vegetation types (Table 3).

A significant finding of the current study was the relatively strong relationship between SOC and above-ground biomass in

the native vegetation. Similar relationships have been found in other ecosystems (e.g. Laurance et al., 1999) but are not commonly reported for the sub-tropics. Unfortunately it is uncertain as to whether: (i) the initial high SOC levels support the potential for greater biomass accumulation; or (ii) whether the greater biomass stocks and associated higher site productivity (e.g. due to rainfall) have contributed to SOC accumulation over a long period of time. Nevertheless, it appears that SOC levels in native vegetation could be a useful predictor (along with other confounding site productivity variables) of potential tree plantation growth in an adjacent area. This is to be expected given the importance of soil organic matter, particularly on sandy soils, in influencing soil productivity variables like cation exchange capacity, microbial biomass and physical properties that can increase soil moisture retention (Reeves, 1997; Oorts et al., 2003; Jia et al., 2005; Lal, 2006; Kimetu et al., 2008).

Litter C, although a relatively small pool (~4.9% of the total ecosystem C in the native and 7.4% in the pine forest) responded significantly to plantation age, with a C sequestration rate of 0.67 Mg ha^{-1} $year^{-1}$. The relationship between litter C and tree basal area for all sites (plantation and native vegetation), although significant, was not strong and hence caution is needed in predicting litter C from basal area measures alone. Further work is needed to determine the best model and combination of variables for litter C prediction, particularly in multi-aged native vegetation. Based on previous studies (Cuevas et al., 1991; Prescott, 2010) we expected the litter C pool to be higher in the *Pinus* plantations than the adjacent native vegetation. Across all sites this hypothesis was partially supported, at least for the fine litter C pool, reflecting the slower decomposition rates of pine needles (Paul and Polglase, 2004; Prescott, 2010). However, this was not the case at all sites, as at one site, fine litter C was higher in the native vegetation than the pine plantation. At this site, the plantation was relatively young (9.7 years since planting) and hence above-ground biomass had not reached levels similar to those in the native vegetation, presumably resulting in lower litter-fall rates. There is a high degree of variation in litter biomass stocks in both plantation and native vegetation, not only in relation to plantation age and basal area, but also to recent site management, such as fire. The finding that understorey woody plant C stocks were higher in the native vegetation than in the pine forest might reflect the inhibitory effect of pine needle litter on native understorey plants (Baker and Murray, 2012), and may also be related to the occurrence of frequent low-intensity fire in the pine forest understories.

4.3. Conclusions

There is regional and global concern regarding the clearing or conversion of native vegetation. While we do not advocate clearing of remnant native vegetation, our findings suggest that historic conversion from native vegetation to commercial plantation has had little influence on ecosystem C stocks across multiple sites in our study region. The two main C pools were above-ground tree C and SOC and neither of these pools was significantly influenced by vegetation type change across all sites. Losses of C from the plantation forests associated with harvesting and plantation management (e.g. site preparation) are offset by increases in live tree biomass and debris pools over time. A key finding of this study is that site-to-site variation in SOC, even within a relatively small geographic area, is much greater than that associated with a change from native vegetation to *Pinus* plantation forest. Another key finding was the significant positive relationship between 0–50 cm SOC and above-ground biomass in the native vegetation areas. Further studies are needed to determine if this relationship holds for other ecosystems and regions.

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