5 Article type[†] : Research Article

Influence of climate on individual tree growth and carbon sequestration in native-tree plantings
Running head: Future native-tree plantings

10

Hania LADA¹, Jian D. L. YEN^{2,3}, Shaun C. CUNNINGHAM¹, Katherine E. SELWOOD²,
Phillip FALCKE¹, Jarrod C. HODGSON⁴ and Ralph MAC NALLY^{1,5}

Family names are capitalized

¹ Institute for Applied Ecology, The University of Canberra, ACT 2601, Australia

15 ² School of BioSciences, The University of Melbourne, VIC 3010, Australia

³ ARC Centre of Excellence for Environmental Decisions, The University of Melbourne, VIC
3010, Australia

⁴ School of Biological Sciences, University of Adelaide, South Australia 5005, Australia

⁵ Sunrise Ecological Research Institute (SERI-OG), Ocean Grove 3226 Australia

20

[†] Corresponding author (Ralph.MacNally@gmail.com)

Author contributions: SCC and RM conceived the study. SCC, PF and JCH undertook field
measurements. JDLY led the analyses with contributions from HL, KES and RM. HL led the
writing with significant textual contributions and editing from all other authors.

25

ORCID numbers:

HL (<https://orcid.org/0000-0002-9916-1933>)

JDLY (<https://orcid.org/0000-0001-7964-923X>)

SCC (<https://orcid.org/0000-0002-1447-312X>)

30 KES (<https://orcid.org/0000-0003-1145-9039>)

¹ SCC passed away in September 2016

This is the author manuscript accepted for publication and has undergone full peer review but has not been through the copyediting, typesetting, pagination and proofreading process, which may lead to differences between this version and the [Version of Record](#). Please cite this article as [doi: 10.1111/AEC.12756](https://doi.org/10.1111/AEC.12756)

Abstract

Native ecosystems face challenges of past and on-going human actions, including vegetation clearance and climate change arising from greenhouse-gas emissions. Reforestation is an important tool for sequestering carbon, so we sought to determine how replanted native trees responded to weather, soil conditions and planting characteristics. We measured girth growth of 13 tree species in 19 native mixed-species plantings and one remnant in south-eastern Australia, bimonthly from 2011 to 2016; replantings ranged between 6–46 years at the commencement of measurements. Band dendrometers (flexible bands that record changes in girth) were used to measure growth, with 34 measurements per tree taken over five years. We used outcomes from models with several plausible weather futures scenarios (Dry, Wet, Wet-to-Dry and Average) for 25 and 50 years for tree girth, and 25 years for carbon accumulation, into the future. Woody species richness enhanced girth growth of all tree species. Higher maximum temperatures and reduced rainfall, which generally are predicted for the region over coming decades, retarded growth of nine tree species. Planting tree density had no discernible association with growth for the range of planting densities used. The most and least carbon was sequestered in Wet and Dry projections, respectively. Three *Acacia* spp. (N-fixers) grew slowest and would sequester least carbon, while four species of *Eucalyptus* grew fastest. These measurements of growth provide critical information for land managers to guide choice in replanting strategies for carbon storage.

Keywords: band dendrometers, carbon sequestration, climate change, reforestation, revegetation

Introduction

The loss and degradation of forests and woodlands arising from human land-use actions is a profound global issue, affecting carbon emissions, biodiversity and ecological processes (Barlow *et al.* 2016, 2018). For example, < 20% of open forest and < 24% of forest landscapes globally were deemed ‘intact’ in 2009 (Potapov *et al.* 2009), and intact forest landscape was reduced by 7.2% between 2000 and 2013 (Potapov *et al.* 2017). Of 62 countries with > 10⁶ ha of forest landscapes, only five had proportions of intact forest landscapes > 50% (Potapov *et al.*

2009). The prevention of further clearances and degradation is a high priority for reducing carbon emissions, as shown by heavy investments in REDD+ (Norman and Nakhooda 2014).

Many forest-landscape regions need reforestation to sequester carbon (C), for biodiversity management and for the provision of ecosystem services (Vesk and Mac Nally 2006; Townsend *et al.* 2012; Giling *et al.* 2013; Ferreira *et al.* 2018).

Reforestation has been used around the world to attempt to reverse the adverse effects stemming from human impacts on forest landscapes. Although ecological restoration

(biodiversity and ecological processes) and C sequestration have been seen as potentially conflicting goals in reforestation (Bekessy and Wintle 2008), there is increasing recognition of the value of jointly pursuing both goals (Polglase *et al.* 2013). For example, the ‘Plant a Billion Trees’ project in the US aims to restore natural ecosystems, to protect remaining forests from invasive species, to sequester C and to improve water quality

(<https://www.plantabillion.org/locations/united-states> ; accessed 15.03.2018). The Australian government plans to plant ‘20 million trees by 2020’ for both environmental conservation and for emissions reduction (<http://www.environment.gov.au/land/20-million-trees>; accessed 15.03.2018). It is important that future climates, warming temperatures and changes in water availability (amounts and seasonal and inter-annual timing) are taken into account in

reforestation strategies given the long time-frames associated with forest regrowth in many parts of the world (Vesk *et al.* 2008).

Mixed-species plantings potentially store at least as much C as monocultures (Hulvey *et al.* 2013), have a greater tree-species diversity, and native, mixed-species plantings have the potential to provide biodiversity benefits alongside C sequestration (Cunningham *et al.* 2015a).

In < 20 years since planting, the amount of C stored in the plant biomass in mixed-species plantings in south-eastern Australia was similar to that of a comparable remnant woodland (Cunningham *et al.* 2015a). Ecological knowledge is critical for establishing plantings that meet the requirements of local species and ecosystem processes (McAlpine *et al.* 2016). Information on the C-storage capacity of tree species, and how these depend upon local conditions (e.g. soil) and planting strategies, is critical for planning by land managers. Although many reforestation activities are conducted as a contribution to ameliorating climate change through C sequestration, climate change itself will affect the growth, survival of individual trees and success of plantings so there is an interaction between emission responses and climate change.

There have been several major assessments of C storage in forests (among other global reservoirs) (Schimel *et al.* 2001; Ballantyne *et al.* 2012). However, there are few high-resolution temporal measurements in field conditions that can link weather and planting conditions to

probable tree growth in future climates. Previous studies have shown correlations between *Eucalyptus* stem growth and weather, but these have been over short periods with small sample sizes (e.g., six trees, one year, Downes *et al.* 1999) or at relatively coarse temporal resolution (e.g., before and after drought-breaking floods, Smith *et al.* 2017).

Knowing how plantings respond to contemporary weather is essential to predict how they will respond to future weather. Moreover, it is important to attempt to account for uncertainties in future climate sequences that are consistent with broader projections of climate changes in a given region (Yen *et al.* 2013). Therefore, we monitored changes in tree girth, which we interpret as growth but that may be negative at times of decreased base xylem pressure potential and soil moisture (Hinckley and Bruckerhoff 1975), in mixed-species plantings in south-eastern Australia. This region is predicted to experience increased temperatures and a redistribution of rainfall among years into long droughts and intense rainfall events (Timbal *et al.* 2015). Using repeated measurements of tree-girth change from mixed-species plantings of different ages and species mixes, we asked:

- (a) How is the growth of each tree species related to weather variation?
- (b) Which planting characteristics (density of trees, woody species richness, presence of nitrogen fixers) and soil characteristics maximize growth and, therefore, C storage of the individual trees of a given species?
- (c) What are the projections for growth by girth and for C storage among different, plausible future climate scenarios for a given species?

Methods

Until 1750, northern and central Victoria (centred on 36.5 °S 146.0 °E) was dominated by forests and woodlands. Plains Grassy Woodlands, Grassy Woodlands, and Box-Ironbark Forests were the most extensive ecological vegetation classes (EVCs) covering 1.94 million ha, and comprising of large *Eucalyptus* trees and a mid-storey of *Acacia*, *Allocasuarina* and small *Eucalyptus* trees (Environmental Conservation Council 2001). European colonization in the 1800s led to widespread clearing for agriculture and for timber harvesting. Only 17% of previously forested and wooded areas retain tree cover, most in the form of post-logging regrowth dominated by small stems, many coppiced, and with some EVCs completely lost (Mac Nally *et al.* 2000, 2002). Extant forests and woodlands have 10–30-m tall open overstorey, and sparse to dense understorey of shrubs (DELWP 2018). Mean rainfall varies spatially from 552 (± 196 temporal SD, based on 30 years) to 931 (± 322) mm yr⁻¹, and mean monthly maximum temperatures range from 12.2 °C in the coldest month to 31.6 °C in the hottest month

(Bureau of Meteorology 2017).

We selected 19 of the 36 mixed-species plantings reported in Cunningham *et al.* (2015a) to provide a representative range of species mixes and planting ages in the region, and we included one native forest remnant for additional data (Fig. 1, Table S1), although this was not intended to be a remnant vs replanting contrast per se. We did not design or implement the plantings but capitalized on variation among plantings to make inferences. All plantings were on previous pastures. All plantings were > 1 ha, non-riparian, fenced from livestock, on yellow sodosol soils, at elevations of 120–370 m above sea level, and were planted by hand from tubestock seedlings into furrows with no other management (Cunningham *et al.* 2015a).

Plantings were 6–46 years old in 2011, dominated by small trees, and with 5–20-m overstorey.

We selected 13 tree species to monitor, 11 of which are dominant in Box-Ironbark Forests and Plains Grassy Woodlands EVCs; one species is indigenous to Western Australia and another to South Australia (Table S2). For each planting, we measured 6–15 trees of each species and 2–4 species per planting (i.e. a total 12–24 trees). If there were > 15 trees of a given species in a planting, we randomly selected 15, although we did not consider saplings and small trees that appeared to be younger than the age of the original planting, as judged by their girth size and height.

Tree measurements

The diameter at breast height (DBH), 1.3 m, of a tree can be used to estimate the above-ground biomass (Bi *et al.* 2004), and hence the C content of a tree, which generally is taken to be 50% of the tree's biomass (Intergovernmental Panel on Climate Change 2003). DBH is a reliable proxy for other components of trees size, including tree height, so that change in DBH is a reasonable measure of tree growth (Bi *et al.* 2004). Band dendrometers measure the expansion of the tree trunk through time and often are used in studies of forest measurements (Liming 1957; Auchmoody 1976). Band dendrometers (Series 5 Manual Band Dendrometer, Agricultural Electronics Corporation) were installed on up to 24 trees at each tree planting in the austral winter of 2011, and at the remnant woodland site in the winter of 2012. We took measurements of the expansion of the trunk every 1–2 months until the austral autumn of 2016. Each dendrometer was fixed to the main stem of the tree at breast height (c. 1.3 m). Each dendrometer had a thin stainless-steel band wrapped around the tree stem and a fixed spring-loaded mount to maintain tension, with a sliding vernier scale to measure girth change. The band dendrometer scale was initially, and after each measurement, set at 5 mm to allow for

possible shrinkage in the tree stem. We converted measurements of girth to diameter using the equation: $\text{diameter} = \text{girth} / \pi$.

We measured the initial and final tree DBH with a diameter tape. We calculated biomass using allometric equations developed by Paul *et al.* (2013), applying the equation for biomass of *Eucalyptus microcarpa* to all tree species that exceeded DBH calibration thresholds for their respective species-specific equations (Table S2). These equations were developed from trees harvested in mixed-species plantings in the area (Paul *et al.* 2013).

Potential predictors

The predictors were weather, soil conditions and planting characteristics. We obtained daily rainfall and maximum daily temperature for the closest weather station (Albury Airport 36.07° S, 146.95° E; Benalla Airport, 36.55° S, 146.00° E; Yarrawonga 36.03° S, 146.03° E; Wallan 37.38° S, 144.97° E; Mangalore Airport 36.89° S, 145.19° E; Redesdale 37.02° S, 144.52° E; Rushworth 36.59° S, 145.02° E) 16.6 km $\pm$ 10 (mean $\pm$ SD) to each planting (Bureau of Meteorology 2017). In five years of repeated measurements, there were both wet (well above-average rainfall) and dry (well below-average rainfall) periods (Bureau of Meteorology 2017). We used 50-m grid data from airborne gamma-ray spectrometric surveys to estimate two soil characteristics: percentage of potassium (K) and the γ Uranium to Thorium ratio, $\gamma\text{U}:\text{Th}$ (Geoscience Australia 2017). Values were extracted for the location of each tree using QGIS (Quantum GIS Development Team 2017). K is an important mineral for plant growth and drought and heat tolerance of plants, while $\gamma\text{U}:\text{Th}$ has negative association with clay content of soil, and so, is an indicator of moisture retention (Pracilio *et al.* 2006).

For each tree and time interval between measurements, we calculated average daily growth as a response variable, and average daily rainfall and average daily maximum temperature as predictors. Other predictors were the tree girth at the time of each previous measurement to ask whether larger trees had faster growth rates, the age of each planting at each measurement (6–52 years old), the presence of nitrogen fixers (N-fixers), woody species richness (all species at a site, and not necessarily only those species for which dendroband measurements were made), and density of trees, total basal area of trees (to see how overall wooded area and thus size of all trees in a planting influences growth of a given species) and type of soil texture (loam, sandy loam or clay loam) of the planting (Cunningham *et al.* 2015a).

Statistical modelling

We used a hierarchical Bayesian model to relate average daily tree growth to the predictor variables. We included separate intercepts and regression coefficients for each tree species and included random intercepts for individual trees, site, year of measurement, and whether the site in which the tree was located was a planting or remnant vegetation. The latter was not strictly a contrast of relevance to the study because there was only one remnant, but this model structure accounted for a potential effect at the single tree level. The hierarchical model structure meant that regression coefficients for each species had a shared hyperprior, which allowed information on much-sampled tree species (e.g., *E. sideroxylon*, Table S1) to inform estimates for more sparsely sampled tree species (e.g., *E. astringens*, Table S1). The model was:

$$g_i \sim \text{Normal}(\mu_i, \sigma_g); \quad (1a)$$

$$\mu_i = \alpha + \gamma_{s(i)} + \mathbf{X}_i \boldsymbol{\beta}_{s(i)} + \kappa_{s(i),h(p(i))} + \delta_{t(i)} + \phi_{p(i)} + \psi_{r(i)} + \rho_{y(i)}; \quad (1b)$$

where: g_i is the i^{th} observation of average daily growth rate, measured for tree $t(i)$ in planting $p(i)$ in year $y(i)$. Tree $t(i)$ belongs to species $s(i)$ and $r(i)$ is a binary variable equal to 1 if tree $t(i)$ were in a planting and 2 if tree $t(i)$ were in remnant vegetation. α is the overall average daily growth rate, $\gamma_{s(i)}$ is the average daily growth rate of species $s(i)$, $\mathbf{X}_i$ is a row vector of predictor variables for observation i , $\boldsymbol{\beta}_{s(i)}$ is a column vector of regression coefficients for species $s(i)$, $\kappa_{s(i),h(p(i))}$ is the effect of soil-texture type h (measured in planting p) on growth rates of species $s(i)$, and δ , ϕ , ψ , and ρ are random intercepts for tree $t(i)$, planting $p(i)$, and year $y(i)$ respectively. The random effects for remnant-status $r(i)$ account for potential systematic, substantial differences in growth rates for individual trees that are in replantings compared to those that are in remnants. All predictor variables except soil texture (a categorical variable) were standardized to zero mean and unit variance prior to analyses. We also ran a model without predictors to obtain the observed daily growth rate of each species.

We assigned a shared hyperprior to the effects of each predictor variable k ($\boldsymbol{\beta}_{\cdot,k} \sim \text{Normal}(\theta_k, \tau_k)$) and assigned exchangeable priors to the random intercepts (δ , ϕ , ψ , and ρ). We used Gaussian priors for all other real-valued parameters, with a mean of zero and standard deviation of two. We used half-Gaussian priors for all variance terms, with a mean of zero and standard deviation of 2.

We fitted the above model in Stan 2.16.0 with the rstan R package (version 2.16.2; Stan Development Team 2017) in R 3.4.0 (R Core Team 2017). For each model, we used three chains each of 20 000 iterations, with the first 10 000 iterations used to ‘tune’ the sampler. We set the target proposal acceptance rate (adapt_delta) to 0.9 and set the maximum tree depth for the NUTS algorithm (max_treedepth) to 30. We considered evidence for a relationship to be

strong, moderate and weak if 95% credible intervals (CIs), 80% CIs and 50% CIs did not overlap zero respectively. Stan and R codes are available at github.com/jdyen/dendrobands.

Future growth and C-storage projections

We ran simulations to predict future growth and C sequestration of each tree species under the scenario of the Representative Concentration Pathway 4.5 (RCP4.5, greenhouse gas emissions peaking by year 2040 then declining, Moss *et al.* 2010; Meinshausen *et al.* 2011) of the Intergovernmental Panel on Climate Change (IPCC), for four plausible scenarios of future rainfall for Murray River Basin (see fractions of dry, average and wet years below). The scenarios were: (1) Dry (based on the outcomes of the CSIRO-MK-3-6-0 model, which predicts mostly dry conditions); (2) Wet-to-Dry (based on ACCESS1.3 model); (3) Average (based on BNU-ESM model); and (4) Wet (based on CESMI-BGC model, which predicts wet conditions) (CSIRO and Bureau of Meteorology 2017). We used these four of the 32 available models to represent a broad range of future climate scenarios for the region. Given that it is uncertain which model is the best, we selected two models with extreme, and two models with average, rainfall projections. It is not possible to get the full range of rainfall scenarios from any one model. We classified recent rainfall data for each season in each year (2000–2016) into dry, average and wet seasons based on 40-year (1911–1950) seasonal averages $\pm 10\%$, (Bureau of Meteorology 2017). For example, a summer was classified as dry if the given summer rainfall (precipitation total from the start of December to the end of February) was $< 90\%$ of the average summer rainfall. This meant that the ranges for average rainfall were 87.3–106.7 mm in summer, 106.8–130.5 mm in autumn, 147.3–180.0 mm in winter, and 124.3–152.0 mm in spring. Between December 2000 and 2016, there were ten dry and four wet summers and autumns, seven dry and four wet winters, and ten dry and four wet springs. The corresponding maximum temperatures for each season in each year 2000–2016 were included in climate data. We classified each of the four future rainfall scenarios (Dry, Wet, Wet-to-Dry and Average) into dry, average and wet years (CSIRO and Bureau of Meteorology 2017) using 40-year annual rainfall averages $\pm 10\%$ as thresholds. The fractions of dry, average and wet years were 0.61, 0.26 and 0.13 respectively in the Dry scenario; 0.53, 0.28 and 0.19 in the Wet-to-Dry scenario; 0.35, 0.4 and 0.25 in the Average scenario; and 0.19, 0.38 and 0.43 in the Wet scenario. We used a Markov chain to sample months randomly, with transition probabilities based on the frequency of transitions to dry, average, and wet years in the rainfall scenarios (CSIRO and Bureau of Meteorology 2017).

We used a first-order Markov model to sample the next month's climate randomly based on current climate: $climate_{j,t+1} \sim \text{Categorical}(\pi_{j' \rightarrow j} \mid climate_{j',t})$; where: $climate_{j,t}$ denotes climate

state j (j = dry, average, or wet) at time t . The parameters $\pi_{j' \rightarrow j}$ are the probability of transitioning to climate state j given current climate state j' .

We used parameter estimates and uncertainties from the Bayesian hierarchical model to project tree girths 25 and 50 years into the future to provide medium and long-term projections.

We started with 6-year old plantings and projected tree girth monthly, ending with 31-year old and 56-year old plantings. We set initial tree girths from values drawn from a random uniform distribution with minima and maxima based on the range of observed, species-specific tree girths in 6–10-year old plantings. We used a Monte Carlo method to incorporate uncertainty into projections, with the following general model form:

$$\begin{aligned} girth_{t+1,sp} = girth_{t,sp} + \\ n_days \times [\alpha + \gamma_{sp} + \beta_{rain} rain_t + \beta_{temp} temperature_t + \beta_{rich} richness_t + \\ \beta_{age} site_age_t + \beta_{size} girth_{t,sp} + \beta_{nfix} n_fixers_present_t]; \end{aligned}$$

where: α , γ and all β s were parameters estimated from the fitted hierarchical model. All model parameters were projected with uncertainties based on posterior distributions from the hierarchical model. $girth_{t,sp}$ is the girth (in mm) of an individual tree of species sp at timepoint t . n_days is the number of days in a given month and was included because parameters in our hierarchical model were estimated for daily tree growth. The variables *rain* and *temperature* were from the climate scenarios. *site_age* is the age of a given planting and was initialized at 6 years and was incremented by one for each subsequent year. *richness* and *n_fixers_present* are woody species richness and an indicator for the presence of N-fixers and were fixed for a given scenario. We calculated tree girth growth for scenarios with 5, 6, 8 and 10 species, and with and without N-fixers (Table S3). We set values of soil K, γ U:Th ratio, tree density and basal area of trees to the average value for all plantings.

Several tree species were only in plantings with > 6 species. Due to the linear, non-saturating structure of our model, extrapolating below or above observed woody species richness values potentially would lead to unrealistic tree sizes. To avoid unreasonable extrapolation, tree-size projections were based on woody species richness within the observed ranges of species richness for each species. This meant that only four species were used in all simulations of woody species richness. The species richness of eight offered the best comparison among species, with only *E. blakelyii* and the non-local *E. cladocalyx* not in plantings with ≥ 8 species.

We simulated 1000 trajectories for each species, climate, woody species richness, and N-fixer scenario (Table S3). We generated a separate climate sequence for each trajectory. Values of rainfall and temperature were drawn from normal distributions with means based on

295 predicted climate scenarios and standard deviations equal to half the range of the observed climate values for that month. We added noise to the observed growth increment (terms involving α , β and γ , above), with mean equal to zero and standard deviation equal to half the predicted growth increment.

We estimated the final above-ground C content from projected tree girth of 31-year old plantings using species-specific equations developed for biomass of trees in mixed-species plantings (Paul *et al.* 2013), and assumed that the C content is 50% of total biomass (Intergovernmental Panel on Climate Change 2003). Species-specific equations were unavailable for *E. albens*, *E. macrorhyncha* and *Al. leuhmanii*, and were inappropriate for DBHs exceeding the range for which the model for a given species was built (Table S2). For these cases, if DBH was < 40 cm we used the generic biomass equation for *Eucalyptus* trees and, if DBH were 40–110 cm, we used the biomass equation for *E. microcarpa* (Paul *et al.* 2013).

Results

The hierarchical model fitted the data moderately well, with $r^2 = 0.46$ for 12 824 observations.

310 There was virtually no effect of a tree being in a planting ($r_{\text{planting}} = 0.017 \pm 0.361$) or in a remnant ($r_{\text{remnant}} = -0.0008 \pm 0.363$).

Tree characteristics and girth growth

The means of average daily girth growth rate were from 0.009 mm day⁻¹ for *Acacia dealbata* to 0.112 for *Eucalyptus astringens*, with strong evidence for growth of *E. astringens*, *E.*

315 *microcarpa*, *E. sideroxylon*, *E. cladocalyx* and *E. leucoxylon*, and moderate evidence for growth of *E. blakelyi*, *E. polyanthemos*, *E. camaldulensis*, *E. macrorhyncha* and *E. albens* because 95% credible intervals (CIs) and 80% CIs, respectively, did not include zero (Fig. 2a). There was moderate evidence that daily growth rates of *Ac. dealbata*, and *Al. leuhmanii* were slower than the growth of *E. astringens* because the 80% CIs did not overlap (Fig. 2a).

320 There was strong evidence for five species and moderate evidence for two species that trees with larger girth in the previous measurement had faster growth rates (Fig. 2b).

Tree mortality

Over five years, 16 individual trees died (= 0.77 % p.a.), all were single-stemmed. Seven deaths were in one planting affected by a local fire (six *Ac. implexa* and one *E. polyanthemos* (Table S1); excluding these deaths, the mortality rate was 0.44 % p.a. Three *E. albens*, one *Al. leuhmanii* and five *Ac. dealbata*, a short-lived species, died of unknown causes. There was no recorded mortality in 14 of the 19 plantings (74%, Table S1).

Weather

There was strong evidence that average daily rainfall was positively associated with the average daily growth of all tree species except for *E. leucoxylon*, for which the evidence was moderate (Fig. 3). There were substantial differences among tree species, with *E. leucoxylon* showing the weakest association with rainfall, followed by *Al. leuhmanii* and *E. blakelyi* (Fig. 3). *E. polyanthemos*, *E. sideroxylon*, *E. cladocalyx*, *E. macrorhyncha* and *E. albens* showed the strongest positive association with rainfall (Fig. 3). Maximum daily temperature was negatively associated with daily growth of nine species of trees (Fig. 3).

Planting characteristics

Planting tree density had no discernible association with growth, but we emphasize that this inference only applies over the range of planting densities we considered (Fig. 3). There was moderate evidence for *E. microcarpa* that total tree basal area of the planting were negatively associated with growth (Fig. 3). While there may seem to be an inconsistency in a negative effect of basal area vs planting density, there was little correlation between these variables ($r = -0.009$), so it is possible for the two predictors to have different relationships with girth growth.

Species richness was the planting characteristic with the strongest positive association with daily girth growth (Fig. 3). Any effects of N-fixers were weak (Fig. 3).

There was little evidence for a relationship between site planting age and daily tree growth, except for *E. sideroxylon*, for which there was weak evidence that growth was slower with age (Fig. 3). Soil texture (clay loam, sandy loam and loam; Fig. 3) had little effect, but there was strong evidence for negative associations with soil K (Fig. 3) and soil γ U:Th (Fig. 3) with growth of 12 and five species respectively.

Tree-girth and C-sequestration projections

Our projections of future tree growth (for trees 6 years old in 2011) after 25 and 50 years with different combinations of species richness and climate (Dry, Wet, Average, Wet-to-Dry) with and without N-fixers suggested that all species grew the most in the Wet scenario and grew the least in the Dry scenario (Figs S1, S2, Table S4). The three N-fixers *Ac. dealbata*, *Ac. implexa* and *Al. leuhmanii* grew most slowly, while *E. polyanthemos*, *E. macrorhyncha*, *E. sideroxylon* and *E. cladocalyx* grew most rapidly, in all the combinations of scenarios of species richness and future climates with and without N-fixers (Tables 1, 2, Tables S4, S5). Consequently, the most and least C per tree was sequestered in Wet and Dry projections, respectively (Fig. S2). All trees grew relatively larger with N-fixers than without N-fixers, except for *E. microcarpa* with species richness of 5 or 6 in the Dry and Wet-to-Dry scenarios, for *E. macrorhyncha*, and

for *A. implexa* with species richness of 8 and 10 in the Dry and Wet-to-Dry scenarios respectively (Table S4).

Discussion

There was strong evidence that greater numbers of woody species in local plantings and greater rainfall enhanced girth growth of the 13 tree species monitored in mixed-species plantings (Fig. 3). Higher maximum temperatures retarded growth of individuals of nine tree species (Fig. 3). Therefore, it is unsurprising that when projecting growth and above-ground C storage with various future climate scenarios and species-richness values, the simulated trees grew the fastest and accumulated most C when in plantings of the highest species richness in the Wet climate scenario, and grew least when species richness was low in the Dry climate scenario (Table S4). Given that species richness has such a strong positive effect, at least up to the maximum richness we observed, it seems advantageous for land managers to plant many different tree species in a given planting. Higher tree-species richness using indigenous species is likely also to increase faunal biodiversity and hence the conservation value of plantings (Mac Nally and Horrocks 2002).

Species richness and N-fixers

Why would species richness have such a positive effect on growth of trees? Potential mechanisms for increased productivity in mixed species plantings include reduced intraspecific competition, resource-use complementarity, reduced damage from disease or pests, facilitation or altered nutrient cycling (Hulvey *et al.* 2013). For example, species less tolerant of high temperatures may benefit from shade provided by taller tree species, which in turn face less competition for sunlight from conspecifics than in a monospecific planting. The roots of different tree species are likely to penetrate to different soil depths, thereby increasing total nutrient-use (Malezieux *et al.* 2009). Mixed forests tend to have much reduced herbivory compared to monocultural forests, probably due to their sheltering more parasitoids or predators of herbivores and the lower availability of host plants, especially for host-specific herbivores (Jactel and Brockerhoff 2007).

The presence of N-fixers in mixed species plantings has been suggested as one mechanism for engendering higher growth rates than in monocultural plantings (Hulvey *et al.* 2013). We found weak effects of the presence of N-fixers in the mixed-species plantings on the growth of eucalypts, suggesting that the primary driver of increased growth rates in mixed species plantings is species richness or species composition, rather than the presence of N-fixers. This may be due to the presence of N in the soil from past applications of fertilizer and

395 overflows from current applications in surrounding pastures (Garten and Ashwood 2002), or
that the densities of N-fixers was too small to much influence rates. However, in our
simulations of future tree growth all species grew larger with N-fixers than without them,
except for *Eucalyptus microcarpa* in plantings with low species richness and for *E.*
macrorhyncha in the Average climate scenario. Even weak associations between N-fixers and
400 growth rates may have a substantial effect over decades.

The fastest growth rate recorded in our study was for *E. astringens*, corresponding to a
3.8 cm increase in DBH annually, which is less than the annual DBH growth of 6 cm for *E.*
camaldulensis recorded between 2008 and 2012 after floods (Smith *et al.* 2017). However, we
did not have data to model the potential effects of herbivory and diseases, so that our long-term
405 projections of continuous growth have resulted in what might be potentially unrealistic tree
sizes of DBH in 56 year-old trees, ranging from 34.5 cm in *Al. leuhmanii* to 189.6 cm in *E.*
polyanthemos in plantings with species richness of 8 under the Wet scenario (Table S4).

Management implications

Land managers establishing mixed-species planting cannot control the weather or future
410 climates but they can choose the mix of tree species, and the locations of plantings and densities
of trees. In the future projections, tree species that grew least in all climate scenarios, Dry, Wet-
to-Dry, Average and Wet, were the three N-fixers, *Al. leuhmanii*, *Ac. dealbata* and *Ac. implexa*.
Al. leuhmanii, which has the hardest wood recorded in the world (Morlan 2017), had the
smallest girth changes and least above-ground carbon. Individuals of *E. polyanthemos* had the
415 largest girth in all simulations, while individuals of *E. macrorhyncha* and *E. sideroxylon* were
projected to grow relatively well under all scenarios. Temperatures are expected to continue to
increase in south-eastern Australia, but there is some uncertainty about future rainfall patterns
both among years and seasonally (Timbal 2015). The species with least variation in growth
among different climate scenarios was *E. leucoxylon* (Table S4).

420 We found little effect of planting density on girth growth of trees and moderate evidence
for negative effects of total basal area on girth growth. The latter may be a better measure of the
intensity of resource competition than density per se. While there was little correlation between
planting tree density and total basal area, plantings with the largest wooded basal areas had on
average the largest trees (Table S2). The low observed mortality rates over the five years of
425 measurements, with some trees killed by a rare fire event in the region, suggest that if such low
rates continue, one might make reasonable estimates of areal rates of C-sequestration by
summation of the individual growth rates and known planting densities, at least up to the
densities we considered.

There were negative effects of γ K and γ U:Th on growth of trees (Fig. 3), which is surprising because K is required for plant growth and heat tolerance. It is possible that native species adapted to soil conditions with lower K are at a competitive disadvantage with weeds when K increases. This negative relationship became evident if soil type were included in the analyses. The negative effects of γ K and γ U:Th indicate that remotely sensed data may be useful in selecting areas to be planted, with areas with lower values of γ K and γ U:Th to be preferred.

Our results suggest that mixed-species planting will be important for C sequestration, at least in our region, with species richness being an important predictor of tree growth rates. If C storage is a primary aim, then including trees such as *E. polyanthemos*, *E. macrorhyncha* and *E. sideroxylon* in mixed plantings may help to maximize carbon sequestration through the high growth rates of individual trees, in place of, or alongside the slower growing *E. albens*, *Al. leuhmanii*, *Ac. dealbata*, and *Ac. implexa*. However, we did not have data to model the potential limiting effects of herbivory and diseases, and so, our long-term projections of continuous growth resulted in probably unrealistic tree sizes of DBH in 56 year-old trees ranging from 34.5 cm in *Al. leuhmanii* to 189.6 cm in *E. polyanthemos* in plantings with species richness of 8 under the Wet scenario (Table S4). In C accounting, agricultural systems are considered to be short-term and of low quality but if these are converted into semi-natural systems for decades, then their ecological value is likely to increase (Ajani *et al.* 2013).

Some plantings may be intended to address biodiversity needs primarily (Ferreira *et al.* 2018), but it is likely that maximizing rates of accrual of C storage at individual planting sites, using mixes of native tree species, is likely to be beneficial if not absolutely optimal for biodiversity too (Vesk *et al.* 2008; Hatanaka *et al.* 2011). If planting is designed to address biodiversity requirements, then the value of species for provision of food sources and shelter for fauna will also need to be considered. For example, seeds of *Al. leuhmanii* are a major food source for the endangered South-eastern Red-tailed Black Cockatoo *Calyptorhynchus banksii graptogyne* (DSE 2006), and flowering eucalypts trees provide food for nectarivorous insects, birds and mammals (Timewell and Mac Nally 2004; Bennett *et al.* 2014; Hawkins *et al.* 2018).

Conclusions

The amount of replanting currently in place is tiny compared with the amount of forest that has been lost. For example, > 11 billion trees had been lost in the Murray-Darling Basin (Australia's largest river system, c. 14% of Australia's land area) in 1993 estimates (Walker *et al.* 1993), so the '20 million trees by 2020' project in Australia is hardly an ambitious target. More

ecologically effective targets, perhaps restoring degraded forest landscapes to > 50%, or $\geq$ 80%, which is mandated in the Brazilian Forest Code (Soares-Filho *et al.* 2010), would have major impacts on C sequestration as trees mature.

The outcomes of our analyses of growth rates of individual trees in sites could form a part of designs for ‘landscape reconstruction’ for biodiversity (Thomson *et al.* 2009). Such reconstruction takes into account landscape connectivity and heterogeneity, and each species’ requirements through space and time to persist in landscapes (Vesk *et al.* 2008). The establishment of significant amounts of sequestered C as quickly as possible using species-rich native-tree replantings would have positive outcomes for biodiversity and improvements in many ecological processes (Bennett *et al.* 2010). Results from our study may be used to inform site-based decisions on planting structure (e.g. combination of species planted) within the context of landscape-scale plans. Where appropriate, information on C-sequestration potential (e.g. soil structure and future climates) could be incorporated in spatial planning for landscape reconstruction alongside project vegetation requirements of species (Cunningham *et al.* 2015b; McAlpine *et al.* 2016).

Acknowledgements

We thank Patrick Baker, Ross M. Thompson, Tim R. Cavagnaro and Philip De Zylva for their earlier involvement with the project, and land owners for access to their properties. All co-authors deeply regret the passing of SCC in September 2016, who established and managed this project.

Funding: This study was supported by ARC Linkage Project and Partner Organizations Parks Victoria, Department of Environment, Land, Water and Planning, Goulburn Broken CMA, North Central CMA and Victorian EPA [grant no. LP120200217]. JDLY was supported by the ARC Centre of Excellence for Environmental Decisions [grant no. CE11001000104].

Literature Cited

Ajani J. I., Keith H., Blakers M., Mackey B. G. & King H. P. (2013) Comprehensive carbon stock and flow accounting: A national framework to support climate change mitigation policy. *Ecol. Econ.* **89**, 61–72.

Auchmoody L. (1976) Accuracy of band dendrometers. (ed USDA). Forest Service, U.S. Department of Agriculture, Upper Darby.

Ballantyne A. P., Alden C. B., Miller J. B., Tans P. P. & White J. W. C. (2012) Increase in observed net carbon dioxide uptake by land and oceans during the past 50 years. *Nature* **487**, 70–2.

- 495 Barlow J., Lennox G. D., Ferreira J., Berenguer E., Lees A. C., Mac Nally R., Thomson J. R.,
Ferraz S. F. D. B., Louzada J., Oliveira V. H. F., Parry L., Ribeiro De Castro Solar R.,
Vieira I. C. G., Aragaõ L. E. O. C., Begotti R. A., Braga R. F., Cardoso T. M., Jr R. C. D. O.,
Souza C. M., Moura N. G., Nunes S. S., Siqueira J. V., Pardini R., Silveira J. M., Vaz-De-
Mello F. Z., Veiga R. C. S., Venturieri A. & Gardner T. A. (2016) Anthropogenic
500 disturbance in tropical forests can double biodiversity loss from deforestation.
Nature **535**, 144-7.
- Barlow J., França F., Gardner T. A., Hicks C. C., Lennox G. D., Berenguer E., Castello L.,
Economio E. P., Ferreira J. & Guénard B. (2018) The future of hyperdiverse tropical
ecosystems. *Nature* **559**, 517.
- 505 Bekessy S. A. & Wintle B. A. (2008) Using Carbon Investment to Grow the Biodiversity
Bank. *Conservation Biology* **22**, 510-3.
- Bennett J. M., Clarke R. H., Thomson J. R. & Mac Nally R. (2014) Variation in abundance of
nectarivorous birds: Does a competitive despot interfere with flower tracking?
Journal of Animal Ecology **83**, 1531-41.
- 510 Bi H., Turner J. & Lambert M. J. (2004) Additive biomass equations for native eucalypt
forest trees of temperate Australia. *Trees* **18**, 467-79.
- Bureau of Meteorology. (2017) Climate Data Online. Bureau of Meteorology (Australia).
CSIRO & Bureau of Meteorology. (2017) Climate Change in Australia website Available
from <http://www.climatechangeinaustralia.gov.au/>.
- 515 Cunningham S. C., Cavagnaro T. R., Mac Nally R., Paul K. L., Baker P. J., Beringer J., Thomson
J. R. & Thompson R. M. (2015a) Reforestation with native mixed-species plantings
in a temperate continental climate effectively sequesters and stabilizes carbon
within decades. *Global Change Biol.* **21**, 1552-66.
- Cunningham S. C., Mac Nally R., Baker P. J., Cavagnaro T. R., Beringer J., Thomson J. R. &
520 Thompson R. M. (2015b) Balancing the environmental benefits of reforestation in
agricultural regions. *Perspectives in Plant Ecology, Evolution and Systematics* **17**,
301-17.
- DELWP. (2018) Bioregions and EVC benchmarks. Available from
[https://www.environment.vic.gov.au/biodiversity/bioregions-and-evc-](https://www.environment.vic.gov.au/biodiversity/bioregions-and-evc-benchmarks)
525 [benchmarks](https://www.environment.vic.gov.au/biodiversity/bioregions-and-evc-benchmarks).
- DSE. (2006) Action Statement. South-eastern Red-tailed Black-Cockatoo *Calyptorhynchus*
banksii graptogyne. Available from

https://www.environment.vic.gov.au/_data/assets/pdf_file/0024/32883/Red-tailed-Black-Cockatoo-Calyptorhynchus-banksii-graptogyne.pdf.

- 530 Environmental Conservation Council. (2001) Box-ironbark. Forest and woodland investigation. Final report. Environmental Conservation Council (Victoria), Melbourne.
- Ferreira J., Lennox G. D., Gardner T. A., Thomson J. R., Berenguer E., Lees A. C., Mac Nally R., Aragão L. E. O. C., Ferraz S. F. B., Louzada J., Moura N., Fonseca Oliveira V. H., Pardini R., de Castro Solar R. R., Vieira I. C. G. & Barlow J. (2018) Carbon-focused conservation may fail to protect the most biodiverse tropical forests. <https://doi.org/10.1038/s41558-018-0225-7>.
- 535 Garten C. & Ashwood T. (2002) Landscape level differences in soil carbon and nitrogen: Implications for soil carbon sequestration. *Global Biogeochemical Cycles* **16**, 61-1--14.
- 540 Geoscience Australia. (2017) The Radiometric Map of Australia. Geoscience Australia.
- Giling D. P., Grace M. R., Mac Nally R. & Thompson R. M. (2013) The influence of native replanting on stream ecosystem metabolism in a degraded landscape: Can a little vegetation go a long way? *Freshwater Biology* **58**, 2601-13.
- 545 Hatanaka N., Wright W., Loyn R. H. & Mac Nally R. (2011) 'Ecologically complex carbon'-linking biodiversity values, carbon storage and habitat structure in some austral temperate forests. *Global Ecology and Biogeography* **20**, 260-71.
- Hawkins B. A., Thomson J. R. & Mac Nally R. (2018) Regional patterns of nectar availability in subtropical eastern Australia. *Landscape Ecology* **33**, 999-1012.
- 550 Hinckley T. M. & Bruckerhoff D. N. (1975) The effects of drought on water relations and stem shrinkage of *Quercus alba*. *Canadian Journal of Botany* **53**, 62-72.
- Hulvey K., Hobbs R., Standish R., Lindenmayer D., Lach L. & Perring M. (2013) Benefits of tree mixes in carbon plantings. *Nature Climate Change* **3**, 869-74.
- Intergovernmental Panel on Climate Change. (2003) IPCC good practice guidance for Land Use, Land-Use Change and Forestry. (eds J. Penman, M. Gytarsky, T. Hiraishi, T. Krug, K. Dina, R. Pipatti, L. Buendia, K. Miwa, T. Ngara, K. Tanabe and F. Wagner) p. 4.101. The Institute for Global Environmental Strategies (IGES), Japan.
- 555 Jactel H. & Brockerhoff E. (2007) Tree diversity reduces herbivory by forest insects. *Ecol. Lett.* **10**, 835-48.

- 560 Liming F. (1957) Homemade dendrometers. *J. For.* **55**, 575-7.
- Mac Nally R., Bennett A. F. & Horrocks G. (2000) Forecasting the impacts of habitat fragmentation. Evaluation of species-specific predictions of the impact of habitat fragmentation on birds in the box-ironbark forests of central Victoria, Australia. *Biological Conservation* **95**, 7-29.
- 565 Mac Nally R., Bennett A. F., Brown G. W., Lumsden L. F., Yen A., Hinkley S., Lillywhite P. & Ward D. (2002) How well do ecosystem-based planning units represent different components of biodiversity? *Ecological Applications* **12**, 900-12.
- Mac Nally R. & Horrocks G. (2002) Habitat change and restoration: responses of a forest-floor mammal species to manipulations of fallen timber in floodplain forests.
- 570 *Animal Biodiversity & Conservation* **25**, 41-52.
- Malezieux E., Crozat Y., Dupraz C., Laurans M., Makowski D., Ozier-Lafontaine H., Rapidel B., de Tourdonnet S. & Valantin-Morison M. (2009) Mixing plant species in cropping systems: concepts, tools and models. A review. *Agronomy for Sustainable Development* **29**, 43-62.
- 575 McAlpine C., Catterall C. P., Mac Nally R., Lindenmayer D., Reid J. L., Holl K. D., Bennett A. F., Runtz R. K., Wilson K., Hobbs R. J. & Seabrook L. (2016) Integrating plant-and animal-based perspectives for more effective restoration of biodiversity. *Frontiers in Ecology and the Environment* **14**, 37-45.
- Meinshausen M., Smith S., Calvin K., Daniel J., Kainuma M., Lamarque J.-F., Matsumoto K.,
- 580 Montzka S., Raper S., Riahi K., Thomson A., Velders G. & van Vuuren D. (2011) The RCP greenhouse gas concentrations and their extensions from 1765 to 2300. *Clim. Change* **109**, 213-41.
- Morlan J. (2017) Wood Species Janka Hardness Scale/Chart By Common/Trade Name A - J. The World's Top 125 Known Hardest Woods. . Available from
- 585 <https://web.archive.org/web/20120524095944/http://www.morlanwoodgifts.com:80/MM011.ASP?pageno=207>.
- Moss R., Edmonds J., Hibbard K., Manning M., Rose S., van Vuuren D., Carter T., Emori S., Kainuma M., Kram T., Meehl G., Mitchell J., Nakicenovic N., Riahi K., Smith S., Stouffer R., Thomson A., Weyant J. & Wilbanks T. (2010) The next generation of
- 590 scenarios for climate change research and assessment. *Nature* **463**, 747-56.

Norman M. & Nakhooda S. (2014) The State of REDD+ Finance. Centre for Global Development, Washington DC.

Paul K., Roxburgh S., England J., Ritson P., Hobbs T., Brooksbank K., RJ R., JS L., Murphy S., Norris J., Neumann C., Lewis T., Jonson J., Carter J., McArthur G., Barton C. & Rose B. (2013) Development and testing of generic allometric equations for estimating above-ground biomass in mixed-species environmental plantings. *For. Ecol. Manage.* **310**, 483-94.

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. (2013) Potential for forest carbon plantings to offset greenhouse emissions in Australia: economics and constraints to implementation. *Clim. Change* **121**, 161–75.

Potapov P., Hansen M. C., Laestadius L., Turubanova S., Yaroshenko A., Thies C., Smith W., Zhuravleva I., Komarova A., Minnemeyer S. & Esipova E. (2017) The last frontiers of wilderness: Tracking loss of intact forest landscapes from 2000 to 2013. *Science Advances* **3**, e1600821.

Potapov P., Laestadius L., Yaroshenko A. & Turubanova S. (2009) Global mapping and monitoring the extent of forest alteration: The intact forest landscapes method. In: *Forest resources assessment working paper*. Forestry Department, FAO, Rome.

Pracilio G., Adams M., Smettem K. & Harper R. (2006) Determination of Spatial Distribution Patterns of Clay and Plant Available Potassium Contents in Surface Soils at the Farm Scale using High Resolution Gamma Ray Spectrometry. *Plant Soil* **282**, 67–82.

Quantum GIS Development Team. (2017) QGIS Geographic Information System. Open Source Geospatial Foundation Project.

R Core Team. (2017) R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Schimel D. S., House J. I., Hibbard K. A., Bousquet P., Ciais P., Peylin P., Braswell B. H., Apps M. J., Baker D., Bondeau A., Canadell J., Churkina G., Cramer W., Denning A. S., Field C. B., Friedlingstein P., Goodale C., Heimann M., Houghton R. A., Melillo J. M., Moore B. J., Murdiyarso D., Noble I., Pacala S. W., Prentice I. C., Raupach M. R., Rayner P. J., Scholes R. J., Steffen W. L. & Wirth C. (2001) Recent patterns and mechanisms of carbon exchange by terrestrial ecosystems. *Nature* **414**, 169-72.

- Smith R., Renton M. & Reid N. (2017) Growth and carbon sequestration by remnant *Eucalyptus camaldulensis* woodlands in semi-arid Australia during La Nina conditions. *Agricultural and Forest Meteorology* **232**, 704–10.
- Soares-Filho B., Moutinho P., Nepstad D., Anderson A., Rodrigues H., Garcia R., Dietzsch L., Merry F., Bowman M., Hissa L., Silvestrini R. & Maretti C. (2010) Role of Brazilian Amazon protected areas in climate change mitigation. *Proceedings of the National Academy of Sciences* **107**, 10821-6.
- Stan Development Team. (2017) RStan: the R interface to Stan. R package version 2.12.1.
- Thomson J. R., Moilanen A. J., Veski P. A., Bennett A. F. & Mac Nally R. (2009) Where and when to revegetate: A quantitative method for scheduling landscape reconstruction. *Ecological Applications* **19**, 817-28.
- Timbal B., Abbs D., Bhend J., Chiew F., Church J., Ekström M., Kirono D., Lenton A., Lucas C., McInnes K., Moise A., Monselesan D., Mpelasoka F., Webb L. & Whetton P. (2015) Murray Basin Cluster Report, Climate Change in Australia Projections for Australia's Natural Resource Management Regions: Cluster Reports. (eds M. Ekström, P. Whetton, C. Gerbing, M. Grose, L. Webb and J. Risbey). CSIRO and Bureau of Meteorology, Australia.
- Timewell C. A. R. & Mac Nally R. (2004) Diurnal foraging-mode shifts and food availability in nectarivore assemblages during winter. *Austral Ecology* **29**, 264-77.
- Townsend P. V., Harper R. J., Brennan P. D., Dean C., Wu S., Smettem K. R. J. & Cook S. E. (2012) Multiple environmental services as an opportunity for watershed restoration. *Forest Policy and Economics* **17**, 45-58.
- Veski P. A. & Mac Nally R. (2006) The clock is ticking - Revegetation and habitat for birds and arboreal mammals in rural landscapes of southern Australia. *Agriculture, Ecosystems and Environment* **112**, 356-66.
- Veski P. A., Nolan R., Thomson J. R., Dorrough J. W. & Mac Nally R. (2008) Time lags in provision of habitat resources through revegetation. *Biological Conservation* **141**, 174-86.
- Walker J., Bullen F. & Williams B. G. (1993) Ecohydrological changes in the Murray-Darling Basin. 1. The number of trees cleared over two centuries. *Journal of Applied Ecology* **30**, 265-73.

Yen J. D. L., Bond N. R., Shenton W., Spring D. A. & Mac Nally R. (2013) Identifying effective
 water-management strategies in variable climates using population dynamics
 models. *Journal of Applied Ecology* **50**, 691-701.

Figure legends

Figure 1. Locations of native mixed-species plantings surveyed, and shown as black dots. Grey represents native forests, woodlands and rivers, white is for cleared land.

Figure 2. Average daily girth growth rates of tree species, and regression coefficients showing effects of tree girth on own growth in native mixed-species plantings. Black dots are means, thick bars are 80% credible intervals, and the thin bars are 95% credible intervals.

Tree species are *Acacia dealbata* (*Ac.d*), *Ac. implexa* (*Ac.i*), *Allocasuarina leuhmanii* (*A.l*), *E. albens* (*E.al*), *Eucalyptus astringens* (*E.as*; non-native), *E. blakelyi* (*E.b*), *E. camaldulensis* (*E.ca*), *E. cladocalyx* (*E.cl*; non-native), *E. leucoxylon* (*E.l*), *E. macrorhyncha* (*E.ma*), *E. microcarpa* (*E.mi*), *E. polyanthemos* (*E.p*), *E. sideroxylon* (*E.s*).

Figure 3. Regression coefficients indicating the effects of weather and planting characteristics on the average daily girth growth rates of tree species in native mixed-species plantings. Black dots are means, thick bars are 80% credible intervals, and the thin bars are 95% credible intervals. Refer to Fig. 2 for species codes.

Table 1 Results of projections of girth of tree species 56 years after planting growing without N-fixers, based on woody species richness within the observed ranges of species richness for each species. *Acacia dealbata* (*Ac.d*), *Ac. implexa* (*Ac.i*), *Allocasuarina leuhmanii* (*A.l*), *E. albens* (*E.al*), *Eucalyptus astringens* (*E.as*; non-native), *E. blakelyi* (*E.b*), *E. camaldulensis* (*E.ca*), *E. cladocalyx* (*E.cl*; non-native), *E. leucoxylon* (*E.l*), *E. macrorhyncha* (*E.ma*), *E. microcarpa* (*E.mi*), *E. polyanthemos* (*E.p*), *E. sideroxylon* (*E.s*). The symbol ‘ $\leq$ ’ indicates the difference of < 2.4 cm in girth between the two tree species.

Climate	Species	Tree girth
Scenario	Richness	

Dry	8	$A.l < Ac.d < Ac.i < E.l < E.al < E.mi < E.ca < E.as < E.ma < E.s < E.p$
Wet-to-Dry	8	$A.l < Ac.d < Ac.i < E.l < E.al < E.mi < E.ca < E.as < E.ma < E.s < E.p$
Average	8	$A.l < Ac.d < Ac.i < E.l < E.al < E.mi < E.ca < E.as < E.ma < E.s < E.p$
Wet	8	$A.l < Ac.d < Ac.i < E.l < E.al < E.mi < E.ca < E.as < E.ma < E.s < E.p$
Dry	5	$E.l < E.b < E.al < E.mi < E.ca < E.p < E.s$
Wet-to-Dry	5	$E.l < E.b < E.al < E.mi < E.ca < E.p < E.s$
Average	5	$E.l < E.b < E.al < E.mi < E.ca < E.p < E.s$
Wet	5	$E.l < E.b < E.al < E.mi < E.ca < E.s < E.p$
Dry	6	$E.l < E.al < E.mi < E.ca < E.p < E.s < E.cl$
Wet-to-Dry	6	$E.l < E.al < E.mi < E.ca < E.s \leq E.p < E.cl$
Average	6	$E.l < E.al < E.mi < E.ca < E.s \leq E.p < E.cl$
Wet	6	$E.l < E.al < E.mi < E.ca < E.s \leq E.p < E.cl$
Dry	10	$A.l < Ac.d < Ac.i < E.al < E.mi < E.s < E.p$
Wet-to-Dry	10	$A.l < Ac.d < Ac.i < E.al < E.mi < E.s < E.p$
Average	10	$A.l < Ac.d < Ac.i < E.al < E.mi < E.s < E.p$
Wet	10	$A.l < Ac.d < E.al < Ac.i < E.mi < E.s < E.p$

Table 2 Results of projections of carbon content of tree species growing without N-fixers in mixed-species plantings of different woody species richness, 31 years after planting.

Acacia dealbata (*Ac.d*), *Ac. implexa* (*Ac.i*), *Allocauarina leuhmanii* (*A.l*), *E. albens* (*E.al*), *Eucalyptus astringens* (*E.as*; non-native), *E. blakelyi* (*E.b*), *E. camaldulensis* (*E.ca*), *E. cladocalyx* (*E.cl*; non-native), *E. leucoxydon* (*E.l*), *E. macrorhyncha* (*E.ma*), *E. microcarpa* (*E.mi*), *E. polyanthemus* (*E.p*), *E. sideroxydon* (*E.s*).

Climate Scenario	Species Richness	Above-ground carbon content of trees
Dry	8	$A.l < Ac.d < Ac.i < E.l < E.al < E.ca < E.mi < E.as < E.ma < E.s < E.p$
Wet-to-Dry	8	$A.l < Ac.d < Ac.i < E.l < E.al < E.ca < E.mi < E.as < E.ma < E.s < E.p$

Average	8	$A.l < Ac.d < Ac.i < E.l < E.al < E.ca < E.mi < E.as < E.ma < E.s < E.p$
Wet	8	$A.l < Ac.d < Ac.i < E.l < E.al < E.ca < E.mi < E.as < E.ma < E.s < E.p$
Dry	5	$E.l < E.b < E.al < E.ca < E.mi < E.p < E.s$
Wet-to-Dry	5	$E.l < E.b < E.al < E.ca < E.mi < E.p < E.s$
Average	5	$E.l < E.b < E.al < E.ca < E.mi < E.p < E.s$
Wet	5	$E.l < E.b < E.al < E.ca < E.mi < E.s < E.p$
Dry	6	$E.l < E.al < E.ca < E.mi < E.p < E.s < E.cl$
Wet-to-Dry	6	$E.l < E.al < E.ca < E.mi < E.s < E.p < E.cl$
Average	6	$E.l < E.al < E.ca < E.mi < E.s < E.p < E.cl$
Wet	6	$E.l < E.al < E.ca < E.mi < E.s < E.p < E.cl$
Dry	10	$A.l < Ac.d < Ac.i < E.al < E.mi < E.s < E.p$
Wet-to-Dry	10	$A.l < Ac.d < Ac.i < E.al < E.mi < E.s < E.p$
Average	10	$A.l < Ac.d < Ac.i < E.al < E.mi < E.s < E.p$
Wet	10	$A.l < Ac.d < E.al < Ac.i < E.mi < E.s < E.p$

Author Manuscript

aec_12756_f1.jpg

aec_12756_f2.jpg

aec_12756_f3.jpg

Minerva Access is the Institutional Repository of The University of Melbourne

Author/s:

Lada, H;Yen, JDL;Cunningham, SC;Selwood, KE;Falcke, P;Hodgson, JC;Mac Nally, R

Title:

Influence of climate on individual tree growth and carbon sequestration in native-tree plantings

Date:

2019-08-01

Citation:

Lada, H., Yen, J. D. L., Cunningham, S. C., Selwood, K. E., Falcke, P., Hodgson, J. C. & Mac Nally, R. (2019). Influence of climate on individual tree growth and carbon sequestration in native-tree plantings. AUSTRAL ECOLOGY, 44 (5), pp.859-870. <https://doi.org/10.1111/aec.12756>.

Persistent Link:

<http://hdl.handle.net/11343/285747>