2016

# Environmental values and fire hazard of eucalypt plantings

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## Publication Details

Jenkins, M., Collins, L., Price, O., Penman, T., Zylstra, P., Horsey, B. & Bradstock, R. (2016). Environmental values and fire hazard of eucalypt plantings. Ecosphere, 7 (11), e01528-1-e01528-14.

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**Disciplines**
Medicine and Health Sciences | Social and Behavioral Sciences

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# Environmental values and fire hazard of eucalypt plantings

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**Citation:** Jenkins, M., L. Collins, O. Price, T. Penman, P. Zylstra, B. Horsey, and R. Bradstock. 2016. Environmental values and fire hazard of eucalypt plantings. Ecosphere 7(11):e01528. 10.1002/ecs2.1528

**Abstract.** The revegetation of cleared landscapes with woody plants (termed "environmental planting") has the potential to sequester carbon (C), provide habitat, and increase biodiversity and connectivity. These environmental values are potentially offset by an increased fire hazard posed by revegetation. There is a need to understand the influence environmental planting has on landscape fire behavior and to determine how this changes as plantings age. This study examined how environmental values, regenerative capacity, fuel metrics, and potential fire behavior change with time since planting. We assessed 57 sites across the Albury-Wodonga region (New South Wales, Australia). This included a range of environmental planting ages (4–40 yr time since planting), remnants, and pastures. Carbon storage increased with age of planting, with largest C stores found in remnants (105 tC/ha), while habitat complexity plateaued around 20 yr, with no significant difference between moderately aged plantings (14–20 yr), old plantings (>20 yr), and remnants. Modeled rate of fire spread was faster in pastures compared to environmental plantings and remnants. Flame height was slightly higher (0.5–1 m) in pastures than environmental plantings and remnants under a Very High Forest Fire Danger Index (FFDI), but this trend reversed under Extreme and Catastrophic conditions with flame heights greatest in environmental plantings and remnants albeit with slower rates of spread. This research highlights the importance of environmental plantings in the landscape in terms of C storage and environmental values and indicates the perceived hazard associated with rate of spread and flame height may not be justified at or less than Very High FFDI. However, at FFDI greater than Very High fire behavior may be significantly enhanced in environmental plantings and remnants. Further consideration needs to be given to the size and design of plantings and the type of species planted to fully develop an understanding of the complexities of fire risk. This study allows land managers to make informed decisions regarding the values and risks associated with revegetation of cleared landscapes with woody plants.

**Key words:** carbon sequestration; fire hazard; fuel loads; habitat complexity; revegetation; vegetation structure.

**Received** 5 June 2016; revised 11 August 2016; accepted 19 August 2016. Corresponding Editor: Debra P. C. Peters.

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## Introduction

The use of land to yield goods and services alters the structure and functioning of ecosystems, and how ecosystems interact with the atmosphere, with aquatic systems, and with surrounding land (Vitousek et al. 1997). Agricultural expansion has resulted in croplands and pastures becoming one of the largest terrestrial biomes, occupying 40% globally (Ramankutty and Foley 1999, Foley et al. 2005). The impacts of land clearing are substantial, with declines in biodiversity, soil stability, and changes to hydrology that have led to secondary salinization (Yates and Hobbs

1997, Harper et al. 2010). Revegetation of the landscape can be motivated by multiple purposes. Positive environmental and social effects of revegetation have been well documented, including rehabilitation of the landscape (Yates and Hobbs 1997, Harper et al. 2005, George et al. 2012), improving biodiversity (Kavanagh et al. 2005, Michael et al. 2011), C sequestration (Harper et al. 2012, Perring et al. 2015, Summers et al. 2015), and for enhancing forage quality and providing shelter for livestock (Williams 1999). However, the potential negative environmental and social effects of revegetation such as increases in pest animals and weeds, changes in fire regimes, and increased fire hazard are rarely considered (Moreira et al. 2001, Benayas et al. 2007).

Positive and negative outcomes of revegetation are likely to vary over time as the planting matures. Forests sequester more C than agricultural plants primarily because trees have substantially larger biomass and longer life spans (Cunningham et al. 2015). Munro et al. (2009*b*) concluded that revegetation was likely to increase structural diversity and condition over time but was unlikely to gain the floristic diversity of remnant vegetation without deliberate intervention. As vegetation changes over time, the suitability of revegetation for any particular species changes and this will favor some species over others (Vesk and Mac Nally 2006). McElhinny et al. (2006) identified structural attributes of eucalypt forests and woodlands that provided key habitat resources for fauna such as fallen timber, canopy, shrub, and ground cover. Indeed, a review of habitat use by terrestrial fauna highlights the importance of maintaining a mosaic of habitats, as many species require multiple habitats to obtain different resources at different life stages (Law and Dickman 1998). Revegetation can provide this variety of habitats required for foraging, shelter, and nesting, and this is in part due to the development of structural complexity over time.

A shift from a grazing system dominated by grasslands to woody vegetation will alter the amount and structural complexity of fuel available to burn in a wildfire (Moreira et al. 2011). A number of studies have provided evidence of increased fire occurrence in Mediterranean rural areas, mainly due to increased woody cover in areas previously used for agriculture or grazing (Duguy et al. 2007, Pausas and Fernández-Muñoz 2012). However, analysis has shown increased fuel accumulation as a result of increased forested area was not necessarily accompanied by parallel increases in area burned (Moreira et al. 2001). Using simulation modeling, Collins et al. (2015) showed that fire size and intensity were altered by increasing woody vegetation extant across landscapes, although the direction of change was dependent on the landscape context such as pre-revegetation forested vegetation extent and pasture fuel loads.

Increases in environmental values (such as structural complexity) and production (i.e., the amount of biomass/fuel) associated with plantings may present conflicting values. Structural complexity of revegetation, as measured by the cover or abundance of a number of vegetation attributes, increases with planting age (Kanowski et al. 2003, Kavanagh et al. 2005) and some key resources that support biodiversity, such as large logs, dead trees, or ground cover complexity can also be viewed as drivers of fire hazard. Indeed, current models of fire behavior such as the Dry Eucalypt Forest Fire Model (Cheney et al. 2012) and the CSIRO Grassland Fire Spread Model (Cheney et al. 1998) use structural fuel parameters such as near-surface, surface and elevated fuel cover scores, and height to predict rate of spread (ROS) and flame height.

Widespread revegetation through environmental planting is needed to redress declining biodiversity and restore sustainable ecosystems, but this must be integrated with an understanding of the potential risks (e.g., fire) associated with increased woody vegetation in the landscape. The study addressed the following questions: (1) Do environmental values increase with planting age? (2) Does potential fire behavior change as planting mature? and (3) Do plantings with "higher" environmental values also pose a greater fire risk? To address these questions, we used both field surveys and modeling to assess how several measures of environmental value (i.e., aboveground C, habitat complexity, regenerative capacity of trees) and potential fire behavior (ROS, flame length) vary across a range of planting ages as well as adjacent pastures and remnants.

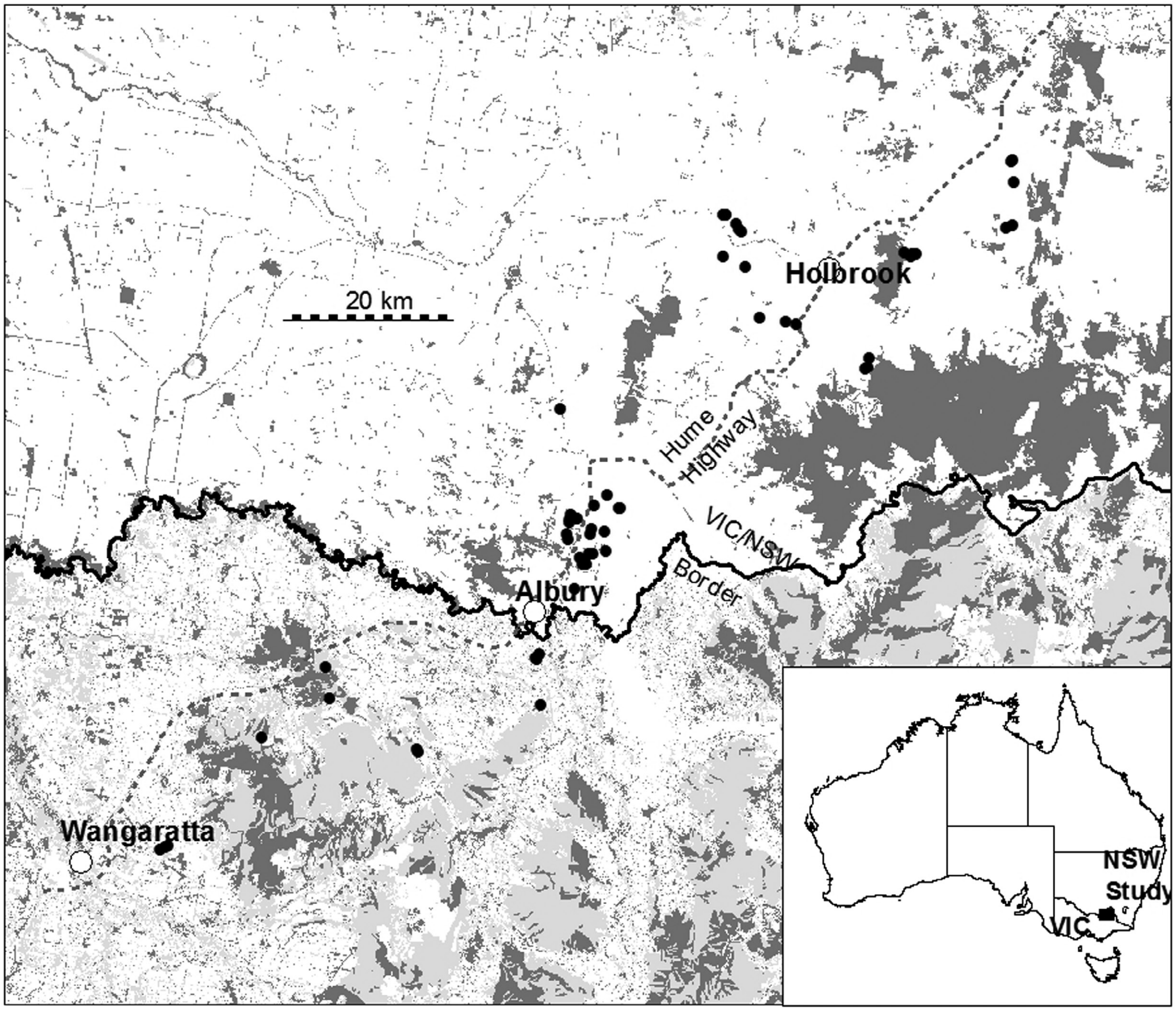


Fig. 1. Map of the study area. Sites are indicated by solid circles ($n$ = 57), and major roads are indicated by dashed lines and New South Wales/Victorian state border by solid line. Shading indicates vegetation type; grassland (light gray) and forest and woodland (dark gray).

## Materials and Methods

### *Study area*

All sites were within the Albury-Wodonga region (36.0806° S, 146.9158° E) in southeastern Australia (Fig. 1). The region has a mean annual rainfall of approximately 618 mm and annual mean maximum temperature of 22.3°C and annual mean minimum of 9.0°C (Albury Airport AWS 072146). Seventy-five percentage of the region has been cleared of native vegetation, and grazing by sheep and cattle is the dominant land use. The remaining area can be classified into three broad vegetation types: Grassy Box-Gum Woodland; Box-Gum Open Forest; River Red Gum Forest and Floodplain Wetlands (Department of Environment Climate Change and Water NSW 2009). From 1977 to 1997, an extensive tree planting program, initiated by the Albury-Wodonga Development Corporation, was conducted, planting a mix of locally indigenous species. Emphasis was placed on putting trees and shrubs into key landscape connections such as creeks and roadsides to create wildlife corridors (G. Datson, *personal communication*). Planting continued after 1997, mostly delivered by local Landcare and community groups and funded by the Natural Heritage Trust and the National

Action Plan for Salinity (K. Durant, *personal communication*).

A range of plantings established between 1974 and 2010 as well as nearby pasture and remnant vegetation were sampled. A total of 57 sites were sampled during September and October 2014, including 12 young eucalypt plantings (PLY; range 3–13 yr), 13 moderately aged eucalypt plantings (range 14–20 yr), 12 old eucalypt plantings (range 22–40 yr), 10 pastures, and 10 remnant patches of vegetation (Fig. 1). Twenty-five tree and eight shrub species were recorded across the study sites. The dominant species were as follows: *Eucalyptus albens* (White Box), *Eucalyptus blakelyi* (Blakely's Red Gum), *Eucalyptus melliodora* (Yellow Box), *Eucalyptus polyanthemos* (Red Box), *Eucalyptus sideroxylon* (Ironbark), and *Acacia dealbata* (Silver Wattle). Aboveground C, habitat complexity, regenerative capacity, and bushfire fuel metrics were measured across each site.

### *Sampling protocol*

*Aboveground carbon*.—A 20 × 50 m (1000 $m^2$) plot was randomly located within each site. In revegetation sites, plots were oriented with planting rows and at least 10 m from the edge of the planting. The species and stem diameter at breast height over bark (dbh; 130 cm) were recorded for all trees with a dbh ≥3 cm. Diameter at breast height was recorded for all stems of multistemmed trees. Diameter at 10 cm height ($D_{10}$) was recorded for all shrubs with a $D_{10}$ ≥3 cm. Counts of stems were recorded for all trees with a dbh <3 cm and shrubs with a $D_{10}$ <3 cm.

Tree and shrub biomass was calculated using generic growth-habit allometric equations for *Eucalyptus* trees and generic shrubs as described by Paul et al. (2013). For plants with multiple stems, we combined the diameters ($D_i$) to give a single value ($D_e = \sqrt{\sum D_i^2}$). For trees and shrubs with a dbh or $D_{10}$ <3 cm (respectively), a constant mass was calculated by applying a dbh or $D_{10}$ of 1.5 cm and using the generic universal equations given in Paul et al. (2013). Carbon mass was then estimated, assuming the C content to be 50% of dry weight biomass (Grierson et al. 1992).

Coarse woody debris (CWD) was sampled using a range of techniques. Standing CWD was assessed across the 20 × 50 m plot area. All standing CWD was measured for dbh (cm), decay class (1–3), and height (m). Carbon was determined using the volume of a cylinder and C content as determined by decay class (Woldendorp et al. 2002, Roxburgh et al. 2006, Table 1).

Table 1. Mean wood densities and C concentrations for coarse woody debris decay classes.

| Wood decay class | Mean density ($g/cm^3$) | Mean C concentrations (%) |
|---|---|---|
| 1 | 0.78 (0.064) | 47.81 (0.14) |
| 2 | 0.70 (0.038) | 48.08 (0.25) |
| 3 | 0.41 (0.035) | 48.00 (0.27) |

*Note:* Mean with standard error shown in parenthesis (modified from Woldendorp et al. 2002, Roxburgh et al. 2006).

Fallen CWD was defined as all debris >0.6 cm in diameter and was determined by two methods. Small CWD (diameter 0.6–10 cm) was sampled using the line intersect method (Van Wagner 1968, Eq. 1). Two parallel 20-m transects were established 10 m apart on the 20 × 20 m subplot. The diameter and decay class (1–3) of each piece of CWD intersecting the line were recorded. The following equation was then used to estimate biomass:

$$M = \frac{\pi^2 \times S}{8L} \sum_i D_i^2 \qquad (1)$$

where $M$ is the biomass (Mg/ha), $D_i$ is the diameter (cm of the piece at the intercept), $S$ is the wood density (Roxburgh et al. 2006, Table 1), and $L$ is the length of the transect line (m). The percentage C in small CWD was calculated based on decay class (1–3) and published C concentrations (Roxburgh et al. 2006, Table 1).

Large CWD (diameter ≥10 cm) was sampled across the 20 × 50 m plot. Each piece of CWD >50 cm long with more than 50% of its length within the plot was surveyed. Large and small end diameter (cm), length (cm), length ≥10 cm diameter, and decay class (1–3) were recorded for each piece of CWD. Volumes of fallen large CWD were estimated using Smalian's formula (volume of a frustum of paraboloid; Eq. 2) as given in Woldendorp et al. (2002).

$$V = L \times \frac{(A_b + A_s)}{2} \qquad (2)$$

$$W = \sum (V \times S) \quad (3)$$

where $V$ is volume of the log ($m^3$), $L$ is piece length (m), $A_b$ is the cross-sectional area at the largest end of piece, $A_s$ is the cross-sectional area at the smallest end of piece. $W$ is the CWD mass (Mg/ha), $S$ is wood density (g/cm$^3$), and mean C concentrations based on decay class (Roxburgh et al. 2006, Table 1). Total aboveground C was determined as the sum of C contained in the trees, shrubs, standing, and fallen CWD per plot.

### Habitat complexity

Habitat complexity is an index of the structural complexity of vegetation and debris, which has been demonstrated to influence the distribution and abundance of fauna (Catling and Burt 1995, Munro et al. 2009*a*, *b*, 2011, Croft et al. 2016). In this study, habitat complexity was determined using a scoring system similar to the one used by Tasker and Bradstock (2006). Scores were allocated to predetermined vegetation categories: canopy, small trees (<2 m), tall shrubs (>2 m), shrubs (1–2 m), small shrubs (<1 m), ground cover, and leaf litter, based on percent cover (Table 2). We included an additional category of fallen CWD (≥10 cm) with a habitat score based on estimated biomass (Table 2) determined by the frequency distribution of the data (mean = 3.35 t/ha, range = 0.02–17.53 t/ha). The scores for each vegetation category were then summed to give a single habitat complexity score for each site.

### Stand characteristics

A 20 × 20 m subplot was established within each 20 × 50 m sampling plot. Tree height (m), height to the base of the canopy (m), and crown width in two directions (m) were recorded for all trees and shrubs located within the subplot. A subsample of the two most dominant eucalypt species per site ($n$ = 3–5 individuals per species) was sampled for regeneration capacity as determined by the presence or absence of fruit during a one-minute search of each tree. Grazing status of each site was assessed by the presence/absence of animals on site or evidence of dung over six 1 × 1 m quadrats. Estimated planting density was recorded and for analysis defined as either low (≤500 stems/ha) or high (>500 stems/ha).

Table 2. Allocated score for habitat complexity, according to percentage estimated cover (canopy, small trees, tall shrubs, shrubs, small shrubs, ground cover, and leaf litter) and measured log mass.

| Score | Cover (%) | Log mass (t/ha) |
|---|---|---|
| 0 | <1 | 0 |
| 1 | 1–5 | ≤0.5 |
| 2 | 6–30 | 0.5–2.5 |
| 3 | 31–70 | 2.51–15 |
| 4 | >70 | >15 |

### Fire characteristics

*Fire behavior predictions.*—Fuel hazard was rated within the 20 × 20 m subplot using the Department of Sustainability and Environment Overall Fuel Hazard Assessment Guide (Hines et al. 2010). Department of Sustainability and Environment hazard ratings for surface, near-surface, elevated, and bark fuel were converted to numeric values 1 (low) to 5 (extreme) and to Dry Eucalypt Forest Fire Model (Vesta) hazard ratings and numeric equivalents (Gould et al. 2008, Hines et al. 2010). Grass fuel loads were estimated using near-surface fuel hazard conversions (Gould et al. 2008).

Rate of spread (km/h) and flame height (m) were predicted using the Dry Eucalypt Forest Fire Model (Vesta; Cheney et al. 2012) for environmental plantings and remnants and the CSIRO Grassland Fire Spread Model (Cheney et al. 1998) for pastures using the measured Vesta hazard ratings and estimated grassland fuel loads. Fuel moisture for all data was calculated using equations given in Gould (1994). Fire behavior was assessed on three levels of the Forest Fire Danger Index (FFDI; McArthur 1967): High/Very High (12–50), Severe (50–75), and Extreme/Catastrophic (>75). For each FFDI, five daily weather observations were obtained from the Australian Bureau of Meteorology, Albury Airport AWS 72146. Five unique weather streams were chosen to allow for variation in the FFDI category and were based on periods when fires actually occurred in the Albury-Wodonga region (Rural Fire Service, *unpublished data*).

### Statistical analysis

Analysis of response variables suggested that relationships with time since planting were nonlinear. Consequently, generalized additive models (GAM, mgcv package; Wood 2011) were used to examine the responses of aboveground C, habitat complexity, regenerative capacity, canopy base height, and fire behavior over time since planting. A range of continuous and categorical predictor variables was used for each response variable. Grazing and planting density were included as factors with presence/absence and low (<500 stems/ha) and high (>500 stems/ha) levels, respectively. Pastures were allocated a time since planting age of zero while remnants were allocated an arbitrary age of 50 yr. All possible additive combinations of time since planting, grazing, and planting density were considered and each model compared using Akaike's information criterion (Appendix S1: Tables S1 and S2). Canopy base height was analyzed by GAM for environmental plantings and remnants to determine the relationship between height and time since planting. All analysis was conducted using R (R Core Team 2015).

## Results

### *Changes in aboveground carbon sequestration and ecological values*

Revegetation sequestered substantially more aboveground C (>0.6 cm diameter) than pastures, with 67.6 ± 7.4 tC/ha stored in older eucalypt plantings (>20 yr; Fig. 2). The largest store of C was in remnant vegetation (105.0 ± 11.1 tC/ha), nearly twice that stored up to 20 yr after revegetation (58.9 ± 8.7 tC/ha) and more than 200 times that stored in pastures (0.44 ± 0.44 tC/ha). A GAM indicated time since planting was a highly significant predictor of aboveground C (Fig. 2; $P < 0.001$, $r^2 = 0.69$), with the presence of grazing having no significant additive effect on aboveground C (Appendix S1: Table S1). The model suggests that sequestration rates are greatest during the first 20 years following planting, after which sequestration rates taper off slightly (Fig. 2).

The best model for habitat complexity included time since planting plus grazing as an additive factor (GAM, $P < 0.001$, $r^2 = 0.61$; Appendix S1: Table S1). Habitat complexity reached a maximum and plateau around 20 yr after planting and was slightly greater in grazed plots than ungrazed (Fig. 3). Grazed plots tended to have higher scores in litter and tall shrub components of the scoring system (Appendix S1: Table S3). The capacity to regenerate (as defined by the presence of fruit) was low in individuals in sites <10 yr since planting and showed a marked increase after 20 yr since planting (Fig. 4).

### *Fire risk modeling*

Fire weather (FFDI) was a key determinant of the nature of the relationship between time since planting and both ROS and flame height. In all scenarios, ROS was greatest in the pastures and ranged from 8.5 km/h at a FFDI of Very High to 13.1 km/h at FFDI of Extreme/Catastrophic (Fig. 5a–c). Modeled ROS were much lower in all sites containing trees and the differences between

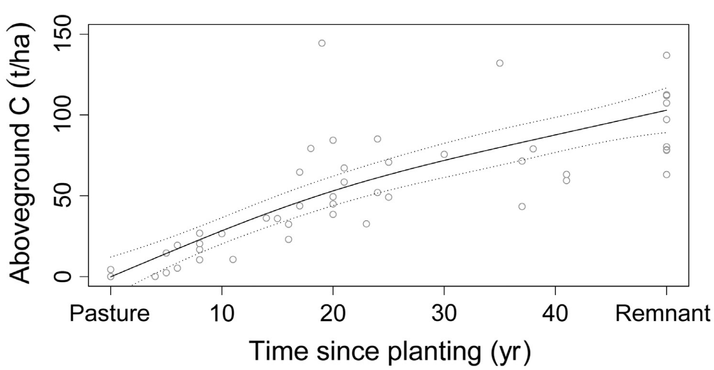


Fig. 2. Aboveground carbon (tC/ha) by time of planting and site type. Aboveground C is a combined value for C contained in tree and shrub biomass, coarse woody debris (CWD), and standing CWD. Solid lines are generalized additive models with 95% confidence intervals indicated by dotted line. Actual data are indicated by the open circles.

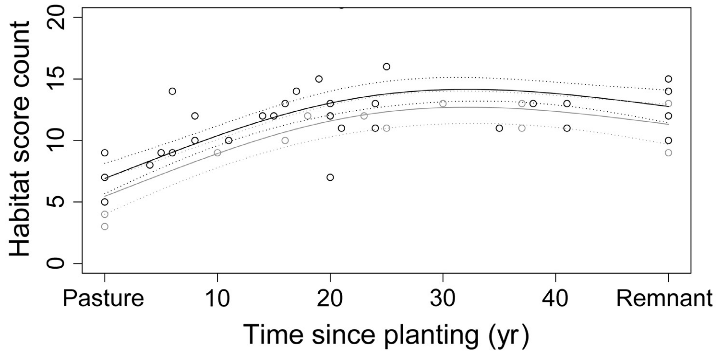


Fig. 3. Habitat complexity score by time since planting and site type. Habitat complexity is a complex score which assess the sites based on structural complexity of vegetation layers according to the percentage estimated cover or mass in each layer (Table 2). Solid lines (black indicating grazed sites, gray ungrazed sites) are generalized additive models with 95% confidence intervals indicated by dotted line. Actual data are indicated by the open circles (black indicating grazed sites, gray ungrazed sites).

the mean ROS for pastures and environmental plantings or remnants increased for Very High (8.0 km/h) to Extreme/Catastrophic (9.6 km/h) fire weather (Fig. 5a–c). However, variability in ROS predictions for plantation and remnants also increased as weather severity increased, and consequently, there was a reduction in the difference between the maximum predicted ROS in pastures and environmental plantings or remnants with increasing fire weather severity (Fig. 5).

Flame height in pastures was on average 0.5–1 m higher in pastures than environmental plantings and remnants under Very High FFDI (Fig. 6a), while under Extreme/Catastrophic

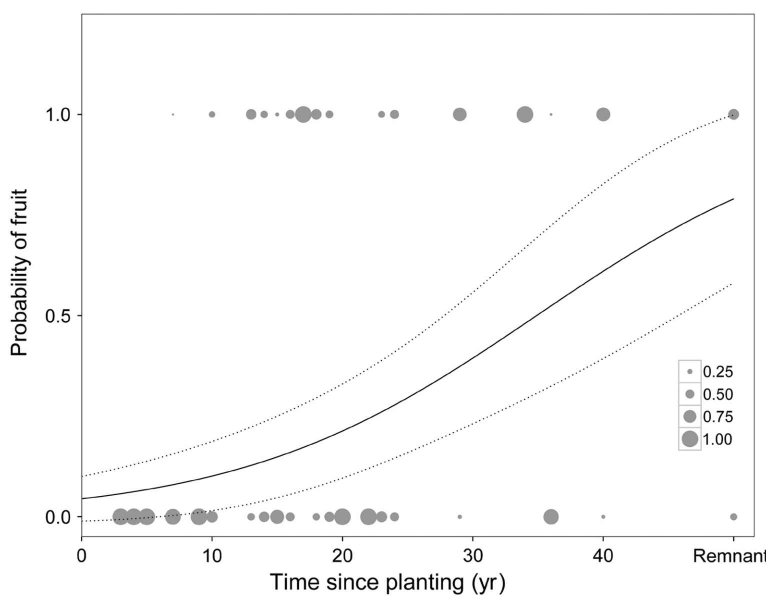


Fig. 4. Regeneration capacity as estimated by the probability of individuals containing fruit by time since planting and site type. Solid lines are generalized additive models with 95% confidence intervals indicated by dotted line. Actual data are indicated by the open circles which represent the proportion of individuals containing fruit.

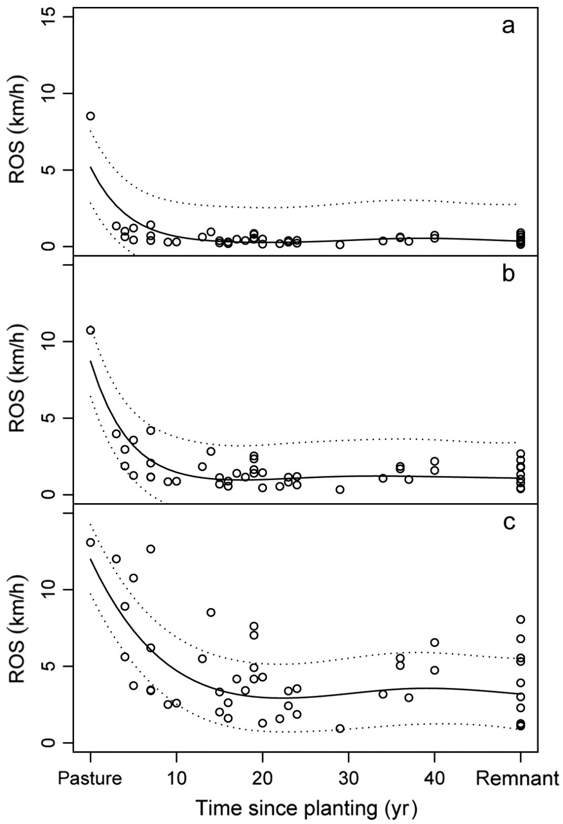


Fig. 5. Model predictions for rate of spread (a–c; ROS, km/h) in response to time since planting (years) and site type, and Forest Fire Danger Index; (a) Very High/High, (b) Severe, and (c) Extreme/Catastrophic. Solid black lines are generalized additive models with 95% confidence intervals indicated by dotted gray lines. Predicted ROS for individual sites are indicated by the open circles.

conditions the reverse occurred with predicted flame heights greatest in environmental plantings and remnants (Fig. 6c). Increasing FFDI from Very High to Severe or Extreme/Catastrophic FFDI resulted in increased modeled flame heights in environmental plantings <10 yr old (Fig. 6a–c). Flame heights in young environmental plantings averaged 11.26 ± 1.37 m at FFDI of Extreme/Catastrophic. Predicted flame height exceeded crown base height at Severe and Extreme/Catastrophic FFDI irrespective of time since planting (Fig. 6b, c). Thus, crown fires would be likely under these conditions in most plantings when crown fuel moisture is sufficiently low.

The nature of the trade-off between fire risk and environmental values is further revealed in bivariate plots showing the mean and distribution of each age group of sites on axes of fire behavior and environmental values (Fig. 7). The introduction of trees into the landscape through environmental plantings (young, middle aged, and older plantings) increased environmental values, and this was coupled with increased flame heights and a decrease in ROS at Extreme/Catastrophic FFDI. Environmental values were lowest in the pastures; however, this could be seen as a trade-off with reduction in fire risk via lower flame heights albeit with greater rates of spread.

## Discussion

Revegetation of the landscape with native woody species had many beneficial outcomes and little evidence of increased potential fire hazard under a FFDI rating of Very High. However, at FFDI ratings above Severe, significantly enhanced fire behavior was predicted. Across the 57 sites examined, environmental values as indicated by aboveground C storage and habitat complexity increased with time since planting. A review of tropical reforestation of abandoned agricultural lands observed similar trends of continuous accumulation of aboveground biomass in forests up to 80 yr of age, with the fastest C accumulation occurring during the first 20 years of regeneration (Silver et al. 2000). Revegetation has the potential to mitigate climate change through sequestration of C. Our results show aboveground C storage in environmental plantings after 20 yr of revegetation (68 tC/ha) was significantly higher than adjacent pasture (0.4 tC/ha) and has the potential to store greater amounts as they age, as indicated by the adjacent remnant vegetation (105 tC/ha). Similar studies of eucalypt plantings grown under low to medium annual rainfall (500–800 mm/yr) reported greater C accumulation: 140.9 tC/ha after 45 yr of revegetation in Eucalypt plantation dominated by *Eucalyptus macrocarpa* (Cunningham et al. 2015), and 117–129 tC/ha after 32–45 yr in plantings of *Eucalyptus cladocalyx* (Paul et al. 2008). In contrast to these studies which showed a mainly linear increase with time, our study clearly indicates a tapering off of biomass accumulation after approximately 20 yr.

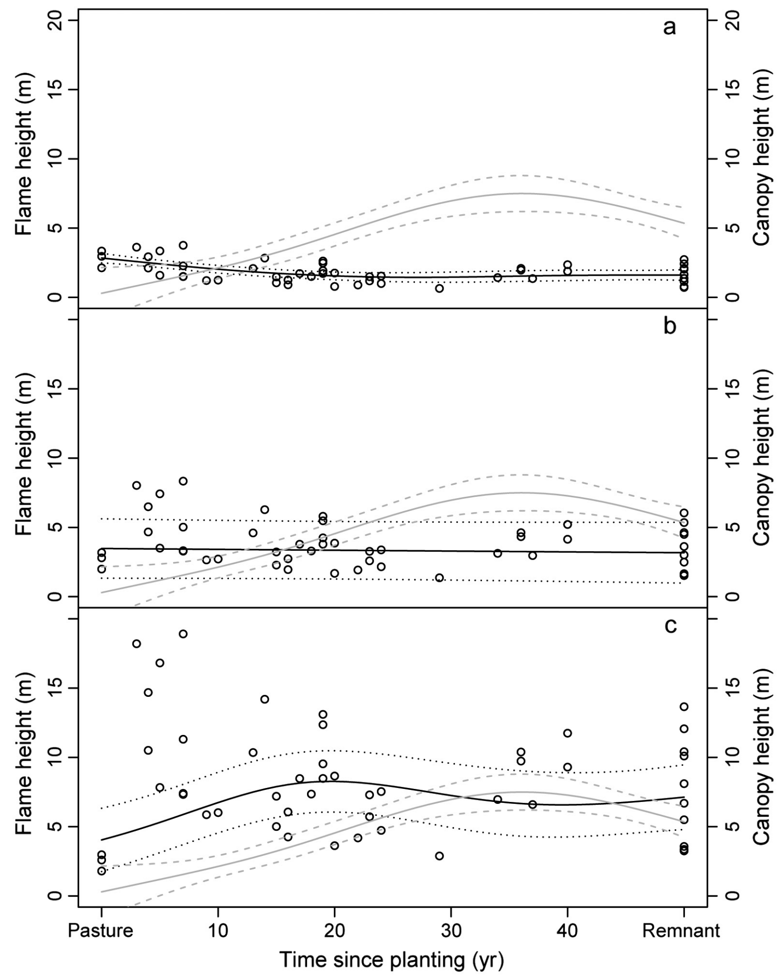


Fig. 6. Model predictions for flame height (a–c; m) in response to time since planting (years) and site type, and Forest Fire Danger Index; (a) Very High/High, (b) Severe, and (c) Extreme/Catastrophic. Solid black lines are generalized additive models with 95% confidence intervals indicated by dotted line. Predicted flame heights for individual sites are indicated by the open circles. Canopy base height (m) is given as solid gray lines (generalized additive models with 95% confidence intervals indicated by dotted gray lines for in environmental plantings and remnants; $P < 0.001$, $r^2 = 0.78$).

Plantations contributed positively to habitat complexity, and this increased with time since planting. Kavanagh et al. (2005) studied plantings in the same region, emphasizing the importance of revegetation for improving biodiversity compared to cleared or sparsely treed paddocks. Our study found habitat complexity developed over time since planting and that by 20 yr after planting, habitat complexity was similar to remnants. This is supported by a study of revegetation sites in tropical and subtropical Australia, which indicated structural complexity tended to increase with age and when left to mature, plantations can develop a similar canopy cover, litter mass, and strata as remnant forest (Kanowski et al. 2003, Munro et al. 2009*b*). A meta-analysis of differences between plantation and pasture indicated birds and reptiles had significantly higher species richness and mammals had higher abundance in plantations than in pasture; however, this response was not observed if remnant vegetation was retained in pasture lands (Felton et al. 2010).

Our study showed eucalypts within environmental plantings were unlikely to seed at

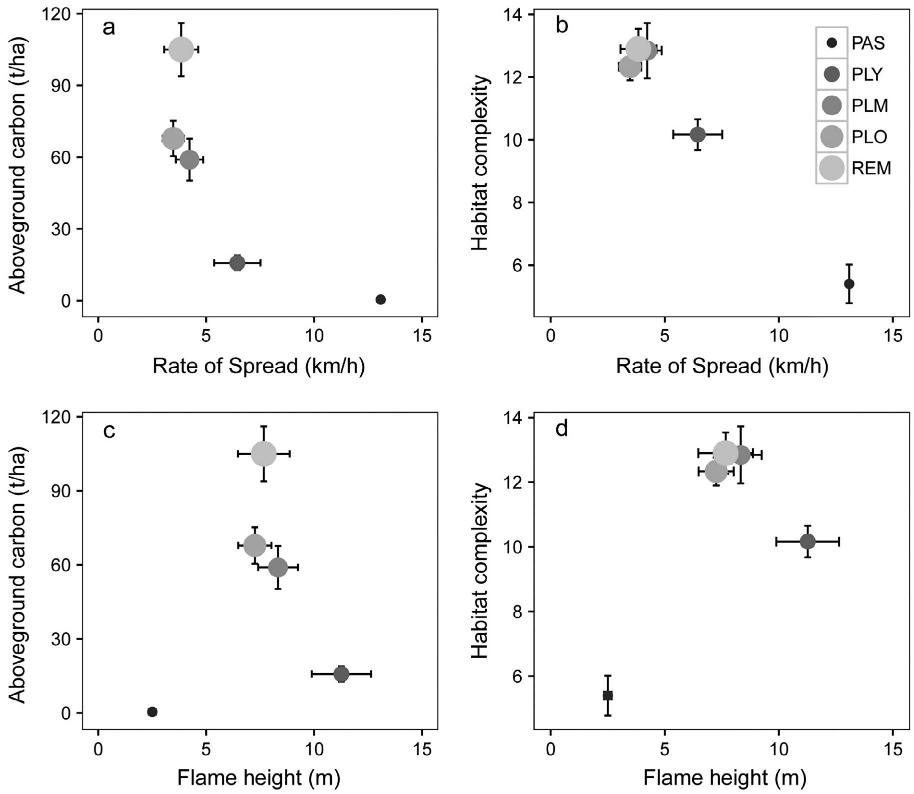


Fig. 7. Trade-off between environmental values and fire behavior at Extreme/Catastrophic Forest Fire Danger Index for pastures (PAS), young eucalypt plantings (PLY), moderately aged eucalypt plantings (PLM), old eucalypt plantings (PLO), and remnant patches of vegetation (REM), mean ± standard error. (a) aboveground C (tC/ha) vs. rate of spread (ROS; km/h), (b) habitat complexity score vs. ROS (km/h), (c) aboveground C (tC/ha) vs. flame height (m), and (d) habitat complexity score vs. flame height (m).

<10 yr since planting, although the presence of fruit increased markedly after 20 yr since planting. This suggests that recruitment within (i.e., infilling) and adjacent to plantings (i.e., expansion) may occur within a couple of decades of planting, provided site factors such as grazing regimes and nutrients are suitable (Fischer et al. 2009). Resprouting eucalypt species dominate Australian temperate woodlands and forests (Gill 1997), such as those examined in our study, and show a tremendous capacity to survive fires via this regeneration mechanism even as seedlings (e.g., Lawes et al. 2011). Pickup et al. (2013) found that the capacity for resprouting eucalypts within environmental plantings <17 yr old to survive fire was high (74–100% survival). Consequently, eucalypt plantings may be regarded as being resilient to fire in the landscape.

Despite all the positive effects increasing trees in the landscape have, a recent survey of landholders indicated that 69% of respondents thought that remnant vegetation and revegetation increase fire risk (Jellinek et al. 2013). Revegetation of agricultural land may enhance landscape surface fuel connectivity, particularly in largely fragmented landscapes, where cleared land may seasonally have very low fuel loads due to cropping and/or grazing (Bradstock 2010). However, agricultural areas in Australia have historically experienced periodic large, fast spreading economically and socially devastating grassfires (Cheney and Sullivan 2008), including several fires in recent years (e.g., in New South Wales in 2002 the Killingsworth fire damaged two houses; in 2006, the Jailbreak Inn fire damaged 11 houses; in 2009, the Wallawalla Rubbish Tip fire damaged five houses; Rural Fire Service, *unpublished data*).

Despite the benefits of environmental plantings in terms of aboveground C storage and habitat complexity, fire behavior in plantings under Severe and Extreme/Catastrophic FFDI poses an elevated threat. Fire behavior predictions showed grasslands have the fastest ROS across all FFDI, whereas young plantings (<5 yr) have a fast ROS and the greatest flame heights under Severe or Extreme/Catastrophic FFDI. Therefore, young environmental plantings may pose the most risk in terms of fire hazard, after which the greater flame heights in the older plantings and remnants pose the most risk due to crowning potential. Zylstra (2013) found that historical fire size in Snow gum was larger in younger fuels and used mechanistic modeling to show that this resulted from fast spreading fires in low grassy regrowth, which slowed slightly as the developing shrubs and trees became taller but were more difficult to suppress due to their large flames. Although the largest flames occurred in the mature forests, these were uncommon because the height of mature trees made them less likely to ignite and more likely to shelter a ground fire from the wind. Zylstra et al. (2016) have shown that the size of this gap between shrub and tree strata is one of the most influential determinants of flame dimensions. Similarly, in eastern Spain, a study of increased fuel accumulation due to land abandonment showed through simulations that the introduction of both dense and open woodlands effectively reduced fire size, while promoting higher biodiversity and landscape resilience against fire (Duguy et al. 2007).

Fire behavior metrics examined in our study were applied at the site scale. However, fire propagation is a complex spatial process and the extent and arrangement of environmental plantings will have important implications for fire risk. To date, there have been few studies addressing this issue. Recent work from southern Australia has shown that while the extent of revegetation can alter fire size and intensity, the direction of change (i.e., increase, decrease, no change) will be dependent on fuel characteristics of surrounding pastures, the extent of native vegetation pre-revegetation and fire weather (Collins et al. 2015). Given the effect of plantings on fire behavior at the site scale, it is likely that the arrangement of plantings will also be of importance for both the spatial characteristics of wildfires (size, ROS, intensity) and ease of suppression. Simulation modeling addressing the importance of planting arrangement will provide valuable insight toward effective planting design at the landscape scale.

One aspect of fire behavior not assessed in our study was ember production (i.e., spotting), which plays an important role in wildfire spread and suppression difficulty (Koo et al. 2010). Bark type will be important in determining spotting potential of trees, with species that have loose fibrous bark or long ribbons having a greater spotting potential than species with smooth or tightly held bark (Hines et al. 2010, Ellis 2011, 2013, Hall et al. 2015). The plantings examined in this study were dominated by box eucalypt species, with bark characterized as short and tightly compacted (Horsey and Watson 2012), and hence unlikely to increase ember production. Consideration of bark types when selecting tree species for environmental plantings will be important in order to minimize potential ember production from plantings. Environmental plantings that avoid eucalypts with stringy and ribbon type bark would decrease the risks associated with ember production.

## Conclusion

Environmental plantings provided greater environmental values, through C sequestration and increased habitat complexity, than adjacent pastures. Plantings offered similar C sequestration potential and habitat complexity after 20 yr since planting to adjacent remnants, highlighting the importance of these plantings. Fire behavior may be significantly enhanced in environmental plantings and remnants under severe and extreme/catastrophic fire weather due to greater flame heights, while across all FFDI pastures posed the greatest risk in terms of rates of spread. These findings show that increased environmental value in plantings is associated with opposing changes in the nature of fire risk (i.e., increased flame height vs. reduced ROS).

## Acknowledgments

We thank the many landholders for allowing us access for fieldwork, Sarah Raymond, Stephanie Samson, Angela Samson for assistance with fieldwork, Brad Law and Traecey Brassil for discussions and data acquisition, Kylie Durrant and Glenda Datson for discussions and additional site history. Michael Bedward for enlightening statistical discussions. This work was funded by NSW Environmental Trust (RD 0178).

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## Supporting Information

Additional Supporting Information may be found online at: http://onlinelibrary.wiley.com/doi/10.1002/ecs2.1528/full