

Article

Impacts of Indigenous Cultural Burning Versus Hazard Reduction on Dry Sclerophyll Forest Composition, Abundance, and Species Richness in Southeast Australia

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Abstract

Fire has had a profound impact on Australia's landscapes and biodiversity since the late Tertiary. Indigenous (Aboriginal) people have lived in Australia for at least 65,000 years and fire is an integral part of their culture and cosmology. In 2015, an Indigenous cultural burn was undertaken by Banbai rangers at Wattleridge Indigenous Protected Area, New England Tablelands, NSW. We compared the impact of this burn on the composition, cover, abundance, and species richness of dry sclerophyll vegetation and fuel hazard, with a hazard reduction burn at nearby Warra National Park, using a Before-After-Control-Impact experimental design. Our study found that the low-severity cultural burn and moderate-severity hazard reduction burn reduced fuel loads but did not have a significant impact on the composition of the vegetation overall or the herb layer. The hazard reduction burn had a significant impact on shrub and juvenile tree (woody species) cover, while the abundance of woody species was significantly affected by both fires, with a mass germination of 'seeder' species, particularly after the cultural burn. The long unburnt fire regime at Wattleridge may have made the vegetation more responsive to fire than the more frequently burnt vegetation at Warra, through accumulation of seed in the seed bank, so that the patchy cultural burn had a greater impact on woody species abundance. In terms of ecological and bushfire management outcomes, this study provides evidence to support claims that Indigenous cultural burning decreases fuel loads, stimulates regeneration of shrubs and trees, and manages at a local, place-based scale. We recommend cultural burning as a key management tool across Indigenous Protected Areas and other land tenures, with its implementation monitored and adaptively managed through two-way science, to foster fire regimes that are both culturally and ecologically beneficial. This is a vital element of our resilience in the Pyrocene and a significant step toward decolonizing science and land management.

Keywords: fire management; cultural fire; Indigenous fire management; protected area management; fire ecology; prescribed fire management; cross-cultural knowledge; Aboriginal rangers



Academic Editor: Grant Williamson

Received: 24 June 2025

Revised: 15 September 2025

Accepted: 16 September 2025

Published: 17 September 2025

Citation: McKemey, M.; Hunter, J.T.; Patterson, M.; Simpson, I.; Reid, N.C.H. Impacts of Indigenous Cultural Burning Versus Hazard Reduction on Dry Sclerophyll Forest Composition, Abundance, and Species Richness in Southeast Australia. *Fire* **2025**, *8*, 367. <https://doi.org/10.3390/fire8090367>

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1. Introduction

Fire has been a major force shaping the distribution and structure of Australian vegetation since the late Tertiary [1]. Ecosystems respond to, and recover from, fire in various

ways, depending on factors such as climate, soil, topography, fire regime, management, and fire–vegetation dynamics [2]. The frequency of fires, as well as their intensity, type, season of occurrence, and extent, are collectively known as the ‘fire regime’ [3], and these have a substantial effect on ecosystems and biodiversity [4], including the composition and structure of dry sclerophyll forests [5]. Fire-response functional traits influence organism persistence across several levels of ecological organization (individual, population, community, landscape). Fire regimes, spatial patterns, and climate influence processes at all levels of organization, and also influence each other through fire weather patterns, fuel accumulation rates, and greenhouse gas emissions [6,7]. Many of Australia’s dominant and most diverse plant genera have traits that allow species to cope with fire as a disturbance, and these traits are likely to have contributed to their success and radiation [2].

For at least 65,000 years, Indigenous peoples and fire have co-existed in the Australian landscape [8–11]. Indigenous groups have managed fire for purposes such as subsistence, ceremony, and communication, and Indigenous cultural fire management continues in Australia today [12–16]. In southeast Australia, a renewal of cultural fire management is underway, with many Indigenous groups re-establishing cultural burning practices on Country (ancestral lands) [17–23]. There are many benefits associated with the renewal of Indigenous cultural fire management in southeast Australia [24]. Ecological benefits include the following: regenerating and restoring ecosystems; maintaining or improving biodiversity values; maintaining micro-climates by protecting the canopy and root systems of plants, and providing management at a local place-based scale [12,18,19,24–30]. Furthermore, cultural burning may provide wildfire management benefits, including reducing the risk of, and destruction caused by, bushfire; reducing fuel loads; and protecting infrastructure [19,24,25,27,28,31–34]. However, few empirical studies have been published to demonstrate these purported benefits [12]. Consequently, Indigenous cultural fire managers have welcomed collaborative research that evaluates the impact of cultural fire management [35,36].

In the New England Tablelands of New South Wales (NSW), the Banbai Aboriginal Nation owns and manages Wattleridge Indigenous Protected Area (IPA). From 2009, Banbai rangers started to reintroduce cultural fire management to their Country. The Banbai rangers, Indigenous fire practitioners and non-Indigenous scientists worked together to understand changes associated with reintroducing cultural burning to a long unburnt ecosystem. Cross-cultural research included monitoring of a cultural keystone species, the short-beaked echidna (*Tachyglossus aculeatus*) [37], and threatened plant species, the Backwater grevillea (*Grevillea scortechinii* subsp. *sarmentosa*) [32], as well as the development of a Banbai fire and seasons calendar [38,39]. Detailed ecological monitoring was undertaken to consider fine-scale changes to vegetation caused by cultural burning, which is the subject of this paper.

Fire is an integral part of the ecology of many communities of the New England Tablelands [40]. In comparison to other regions, New England dry sclerophyll forests are species-rich systems with a large percentage of the plant diversity in the woody shrub layer, including many seeder species [41]. The fire response syndrome of plant species is a key life history trait with profound effects on post-fire population dynamics and community composition [42–48]. Some plant species (‘seeders’) are killed by fire and, after fire, rely on seed germination and a sufficient inter-fire period for seedlings to grow, mature and reproduce, whilst others (‘sprouters’) resprout after fire from root, basal, stem, or apical buds [49–51]. Species can vary in their fire response due to age, herbivory, fire frequency, fire intensity, other plant stresses, soil fertility, climatic events, and ecotypic genetic variation [49]. The response of a species to fire is useful to assess the extirpation risk associated with a given fire regime. Maturation times of new recruits of plants killed by

fire is a critical biological variable in the context of fire regimes, because time to maturation sets the lower limit for fire intervals to avoid local population decline or extirpation of such species [52].

In the New England Tablelands, Clarke, Knox, Campbell, and Copeland [43] found that resprouting was present in 62% of 489 taxa of woody flora, the frequency varying among habitats, growth forms, and growth forms within habitats. Sixty-three per cent of woody plants were resprouters in dry sclerophyll forests. Kitchin [53] found that locations in the New England Tablelands with frequent fire and short inter-fire intervals displayed reduced shrub species richness and abundance, with an accompanying simplification of the structure of the vegetation and reduced shrub abundance in the seed bank. Clarke [40] suggested that fire frequency is even more important than fire response.

Too-frequent fire reduces species richness in all vegetation types, and ‘high frequency fire resulting in the disruption of life cycle processes in plants and animals and loss of vegetation structure and composition’ has been listed as a Key Threatening Process under the NSW *Biodiversity Conservation Act 2016* [54]. On the other hand, many years without fire can lead to the senescence and loss of fire-dependent species [55]. However, these species may persist as long-lived seed banks and not necessarily be lost to the system. The NSW government developed the *Guidelines for Ecologically Sustainable Fire Management* [56], which describes fire-interval domains for broad vegetation groupings and derives fire interval guidelines or ‘thresholds’ based on broad ranging analyses of vital attribute information for vascular plant species known to occur in these groupings.

While many fire ecology studies have been undertaken in southeast Australia, few studies have considered the impact of Indigenous cultural burning on dry sclerophyll forests. This study compared a cultural burn at Wattleridge IPA with a hazard reduction burn implemented by the NSW National Parks and Wildlife Service (NPWS) in a similar ecosystem nearby. We used this case study to explore the potential ecological and bushfire management benefits of cultural burning. Indigenous fire practitioners in southeast Australia generally claim that cultural burning is smaller, patchier, and less severe than other fire types (e.g., hazard reduction burning or wildfire). They assert that the forest canopy is sacred and must not be burnt under Indigenous cultural burning Lore, and that cultural burning leaves habitat for animals, protects special places, encourages regeneration, and helps to prevent destructive wildfires [57–59]. Through this experiment (part of a suite of broader studies, see [58]), we investigate the impacts of cultural burning on vegetation composition, abundance, species richness, and fuel load.

2. Materials and Methods

2.1. Fieldwork

We undertook vegetation monitoring and measured fire indicators pre- and post-fire in a Before-After-Control-Impact (BACI) experimental design [60] at Wattleridge IPA and Warra National Park (NP) (Figure 1). Prior to each fire, 30 monitoring plots were established across the two locations (12 at Wattleridge IPA, 18 at Warra NP). Of the 30 plots, 15 fire treatment plots were placed in areas that were to be burnt according to burn plans [61,62] and 15 control plots were placed in areas to remain unburnt. Based on literature, control and impact plots shared similar vegetation type (New England Dry Sclerophyll Forests; Keith [5]), soil, geology, and climate [63,64]. Plots were randomly placed in each zone. Plot dimensions were 20 × 20 m, as per the NSW Department of Environment Climate Change and Water [65] standard. In each 20 × 20 m plot, there were four nested transects of 5 m² and five quadrats of 1 m².

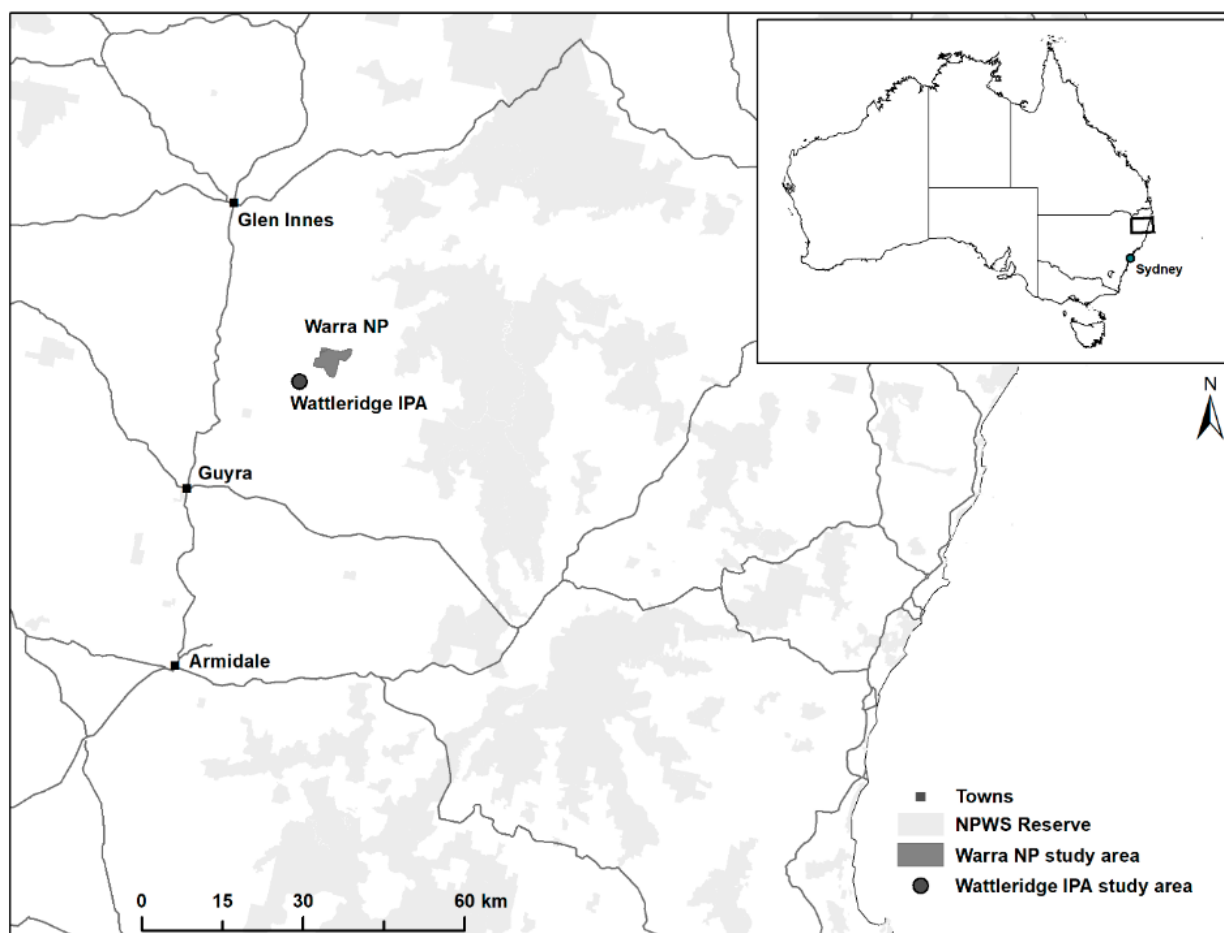


Figure 1. Location of the study area, Wattlebridge Indigenous Protected Area, and adjoining Warra National Park in the New England Tablelands of New South Wales, Australia.

Fire history was ascertained from maps provided by NPWS for Warra NP and the NSW Rural Fire Service (RFS) Prescribed Burn Plan for Wattlebridge IPA [61,62]. Warra NP has a history of frequent large fires while Wattlebridge IPA was comparatively long unburnt prior to 2015 (Table 1).

Table 1. Fire history of vegetation plots in Wattlebridge Indigenous Protected Area (IPA) and Warra National Park (NP), 1990–2015.

	Wattlebridge IPA—All Plots	Warra NP—Impact Plots	Warra NP—Control Plots
Time since last large fire	30 years	16 years	14 years
Number of fires that had affected plots 1990–2015	0	2	3
Total number fires on property 1990–2015	4	10	10
Prescribed fire threshold ¹	Within threshold	Within lower limit of threshold	Too frequently burnt

¹ Fire interval threshold based on vital attributes of plant species in broad vegetation groupings [56].

A cultural burn was undertaken by the Banbai rangers (with RFS support) at Wattlebridge IPA. This fire was a small (4 ha), low-intensity, mosaic burn on 16 August 2015. A hazard reduction burn was undertaken at Warra NP and adjoining private land by NPWS

and RFS. This fire was a large (685 ha) moderate-intensity burn on 20 October 2015. The planned burn area was 351 ha; however, another fire started nearby and joined with the planned burn to increase the total area burnt to 685 ha.

We measured fire indicators in all plots in 2015, 6 weeks before, and 1 year and 3 years after the cultural (Wattleridge) and hazard reduction (Warra) burns. All control plots remained unburned in 2016 and 2018, unlike the fire treatment plots. Fire severity (broadly defined as ecosystem impacts of fire, Keeley [66]) was determined based on the following: fire plans, debrief reports, extent, estimated canopy scorch [61,62], observed intensity, percentage of each plot burnt (percentage bare ground after fire in all quadrats), tree scar height (mean height visually estimated in plots), and change to fuel hazard [67,68].

We identified and estimated the foliage projected cover (cover) [69] of all vascular plant species in the plot, and all herbaceous species in the 1-m² quadrats. In the 5-m² transects, total number (count) of shrubs and juvenile trees (<2 m high) was measured. Data that were subsampled for herbaceous species in the 1-m² quadrats, and for shrubs and juvenile trees in 5-m² transects, were combined and analyzed at the plot level. For each shrub or juvenile tree, the following details were noted: burnt/unburnt, alive/dead, reproductive stage (none/flower/fruit), and fire response (resprouted or germinated from seed after fire). Fire response traits were allocated based primarily on direct observations of species in the post-fire environment, and secondarily from fire responses in Hunter [70,71] and Clarke, Knox, Campbell, and Copeland [43]. Fire response traits for each species are listed in Supplementary Material S2 (Table S1). Plant nomenclature follows that of the PlantNET Information System ([72] accessed 18 September 2020).

2.2. Statistical Analysis

We analyzed 9 response variables (composition–cover and plot species richness of all plant species, of just the herbaceous species, and of just the shrubs and juvenile trees; abundance of all shrub and juvenile tree species, the abundance of just seeder shrub and juvenile tree species, and of just resprouters) in relation to location (Warra NP, Wattleridge IPA), year (2015, 2016, 2018), and fire treatment (control, impact). To assess the effect of fire on the cover of all plant species, of just the herbaceous species and of just the shrub and juvenile tree cover, PERMANOVA analyses were undertaken. Shade plots (visual representations of abundance matrices from multivariate species assemblage studies) were inspected in order to assess which transformations would be most appropriate for each dataset. For multivariate PERMANOVA analysis, square root transformations were used with the full-floristic and herbaceous species cover data, and log ($X + 1$) transformations were used with the shrub and juvenile tree count data, including the abundance of seeders and resprouters. Initial investigation of the data indicated floristic separation due to location (Warra vs. Wattleridge), so PERMANOVA analyses were undertaken separately for each location with plot nested within treatment and year of survey using 999 permutations to test for interactions and pair-wise tests of treatment and year.

To assess the effect of fire on species richness and shrub and juvenile tree abundance, univariate modeling was undertaken using the ‘lm’ package in R [73]. Analyses of variance (ANOVA) were undertaken with both locations together and separately but as the results were similar, only results with locations analyzed together are presented. ANOVA (lm) of full-floristic species richness, shrub species richness and herbaceous species richness was conducted in relation to fire treatment and year, with location as a blocking factor. Post hoc tests (Tukey’s Pairwise Test) in the software package, PAST 4.03 [74], were used to determine differences in full-floristic species richness between location, treatment and year. To assess the effect of fire on shrub and juvenile tree abundance, ANOVA (lm) of the shrub and juvenile tree count data was undertaken in relation to fire treatment and year,

with location as a blocking factor. Similar analyses were repeated for counts of just seeder shrubs and juvenile trees, and of just resprouter shrubs and juvenile trees. One control plot at Wattleridge did not contain any shrubs, so this was removed from analyses of the shrub count data, as it was an obvious outlier. All outputs from PERMANOVA and ANOVA analyses are included in Supplementary Material S1.

3. Results

The fire treatments in 2015 were assessed as low severity at Wattleridge IPA and moderate severity [66] at Warra NP (Table 2). At Wattleridge, canopy trees retained green foliage although the stems were scorched; the surface litter, mosses, and herbs were charred or consumed, while the soil organic layer was largely intact, with charring limited to several millimeters deep. At Warra, some of the tree canopies were killed, but the foliage was not consumed; all understorey plants were charred or consumed; on the soil surface, the fine dead twigs were consumed, logs were charred, and the pre-fire soil organic layer was largely consumed. The cultural burn undertaken by Banbai rangers at Wattleridge was less extensive, patchier, and less intense than the hazard reduction burn undertaken by NPWS at Warra NP (described in detail in [32]).

Table 2. Fire characteristics and severity in impacted plots of the Wattleridge IPA cultural burns (August 2015) and Warra NP hazard reduction burn (October 2015). Data for fuel load decrease, tree scar height, and percentage of plot burnt are mean values $\pm$ 1 standard error (SE) of the mean (n). The mean fuel load decrease and tree scar height did not differ significantly between the two types of fire. Percentage of the plot burnt differed significantly at $p < 0.05$.

Indicator	Wattleridge IPA Cultural Burn	Warra NP Hazard Reduction
Mean fuel load decrease (t/ha)	8.9 ± 1.59 (6)	7.8 ± 1.49 (9)
Mean tree scar height (m)	3.8 ± 0.65 (6)	5.8 ± 1.25 (9)
Mean percentage plot burnt (%)	59 ± 8.0 (6)	81 ± 5.1 (9)
Total fire size (ha)	4	685
Fire plan intensity	Low	Moderate–high
Observed intensity	Low	Moderate
Estimated crown scorch	0%	10%
Fire severity	Light	Moderate

PERMANOVA analyses of the full-floristic cover data were undertaken separately for each location. There was no significant effect of fire on composition (PERMANOVA, Year $\times$ Fire treatment: $p = 0.988$ for Warra, and $p = 0.938$ for Wattleridge, Table 3). However, treatment was significant at Wattleridge (PERMANOVA, $p = 0.046$), indicating a systemic difference between control and treatment plots. There was no evidence of a fire effect on full-floristic species richness (ANOVA, Year $\times$ Fire treatment: $p = 0.914$, Table 4, Figure 2) although, again, treatment was significant at Wattleridge IPA (ANOVA, $p = 0.001$, Table 4).

Herbaceous species richness was higher at Warra NP (mean number of species per plot, 23–26) than Wattleridge (17–24; ANOVA, Location, $p = 7.93 \times 10^{-6}$, Table 4, Figure 3), and Wattleridge fire treatment plots had greater herbaceous species richness (mean of 22–24 species) than control plots (17–20). Fire did not have a significant effect on herbaceous composition at either site (PERMANOVA, Year $\times$ Fire treatment: $p > 0.8$, Table 3) but herbaceous cover in 2015 differed to the composition in 2016 and 2018 at Warra NP across all plots (PERMANOVA, Year: $p = 0.0001$, Table 3). Fires did not have a significant impact on herbaceous species richness (ANOVA, Year $\times$ Fire treatment: $p = 0.2615$, Table 4).

The impact of fire on shrub and juvenile tree species richness was not significant (ANOVA, Year $\times$ Fire treatment: $p = 0.259$). However, the effect of both Year and Location on understorey woody richness was significant (Table 4), with lower shrub and juvenile

tree richness in 2015 than subsequent years and lower richness overall at Warra (mean of 4–10 species per plot) than Wattleridge (7–13 species; Figure 4). The impact of fire on shrub and juvenile tree cover was significant at Warra NP immediately after the hazard reduction burn (PERMANOVA, Year $\times$ Fire treatment: 2015 vs. 2016, $p = 0.002$, Table 3) but not Wattleridge. The following species, which were mostly seeders, increased in abundance: *Bossiaea scortechinii*, *Acacia filicifolia*, *Eucalyptus* sp., *Dillwynia retorta*, *Hovea purpurea*, and *Grevillea scortechinii* subsp. *sarmentosa*. In contrast, *Indigofera australis*, *Bursaria spinosa*, *Leucopogon lanceolatus*, and *Brachyloma daphnoides* declined in abundance and were mostly resprouters. Shrub and juvenile tree abundance increased significantly in the burnt plots at both Warra and Wattleridge (ANOVA, Year $\times$ Fire treatment: $p = 0.0003$, Table 4, Figure 5), particularly 1 year after fire. This was driven by the fire response of seeder species: seeder abundance increased significantly after the fires (ANOVA, $p = 0.002$, Table 4, Figure 6a) whereas resprouter abundance was not significantly affected by fire treatment (ANOVA, $p = 0.2092$, Table 4, Figure 6b). Seeders comprised a greater proportion of shrub and tree species at Wattleridge (24 seeder species [67%] vs. 12 resprouter species [33%]) than at Warra (18 seeders [55%] vs. 15 resprouters [45%]).

Table 3. Summary of the results (p -values) of compositional cover (PERMANOVA) analyses.

Plant Taxa	Location	Year	Fire Treatment	Year $\times$ Fire Treatment
Full floristics	Warra NP	0.175	0.115	0.988
	Wattleridge IPA	0.459	0.046 *	0.938
Herbaceous species	Warra NP	0.0001 ***	0.110	0.828
	Wattleridge IPA	0.087	0.101	0.978
Shrubs and juvenile trees (<2 m high)	Warra NP	0.02 * (2015 v 16 = 0.003 *)	0.031 *	0.353 (Impact plots, 2015 v 16 = 0.002 *)
	Wattleridge IPA	0.983	0.014 *	0.975

* $p < 0.05$, *** $p < 0.001$

Table 4. Summary of the results of univariate (ANOVA) statistical analyses, showing p values for main effects and interactions.

Plant Taxa	Response Variable	Location	Year	Fire Treatment	Year $\times$ Fire Treatment
Full floristics	Plot species richness	0.086	0.734	Overall = 0.005 * (Warra = 0.941, Wattle-ridge = 0.001 *+)	0.914
Herbaceous species	Plot species richness	7.93×10^{-6} ***	0.70349	0.02908 *	0.26153
Shrubs and juvenile trees (<2 m high)	Plot species richness	5.10×10^{-5} ***	0.12998	0.110	0.259
	Abundance	5.786×10^{-10} ***	0.0001 *	9.690×10^{-5} ***	0.0003 ***
	Seeder abundance	7.816×10^{-11} ***	5.339×10^{-5} ***	0.0002 ***	0.0020 **
	Resprouter abundance	0.0132 *	0.3002	+0.0671	0.2092

* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$; +Tukey's pairwise test indicated that control and impact plots differed only in 2015.

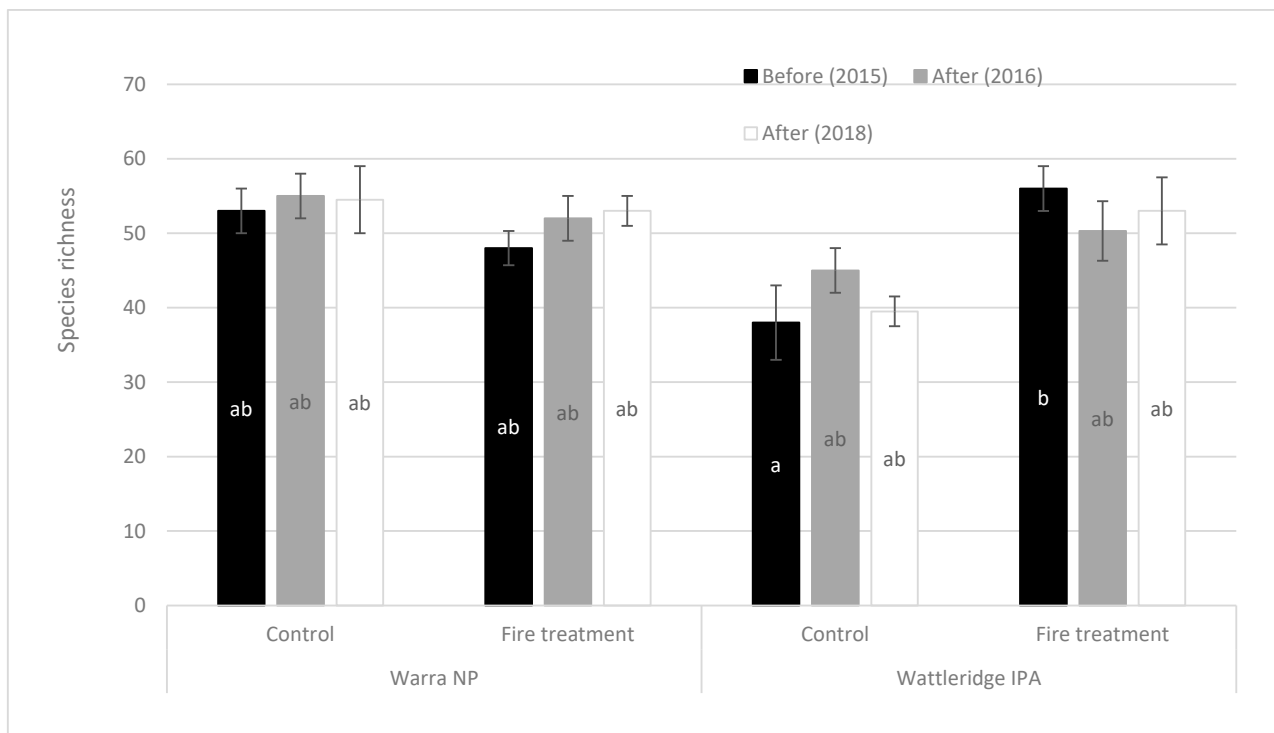


Figure 2. Mean species richness (± 1 SE) of control and fire treatment (impact) plots for full floristics at Warra NP and Wattleridge IPA before (2015) and after (2016 and 2018) treatment (showing Tukey's pairwise groupings, a and b; means not sharing the same letter differed significantly, $p < 0.05$).

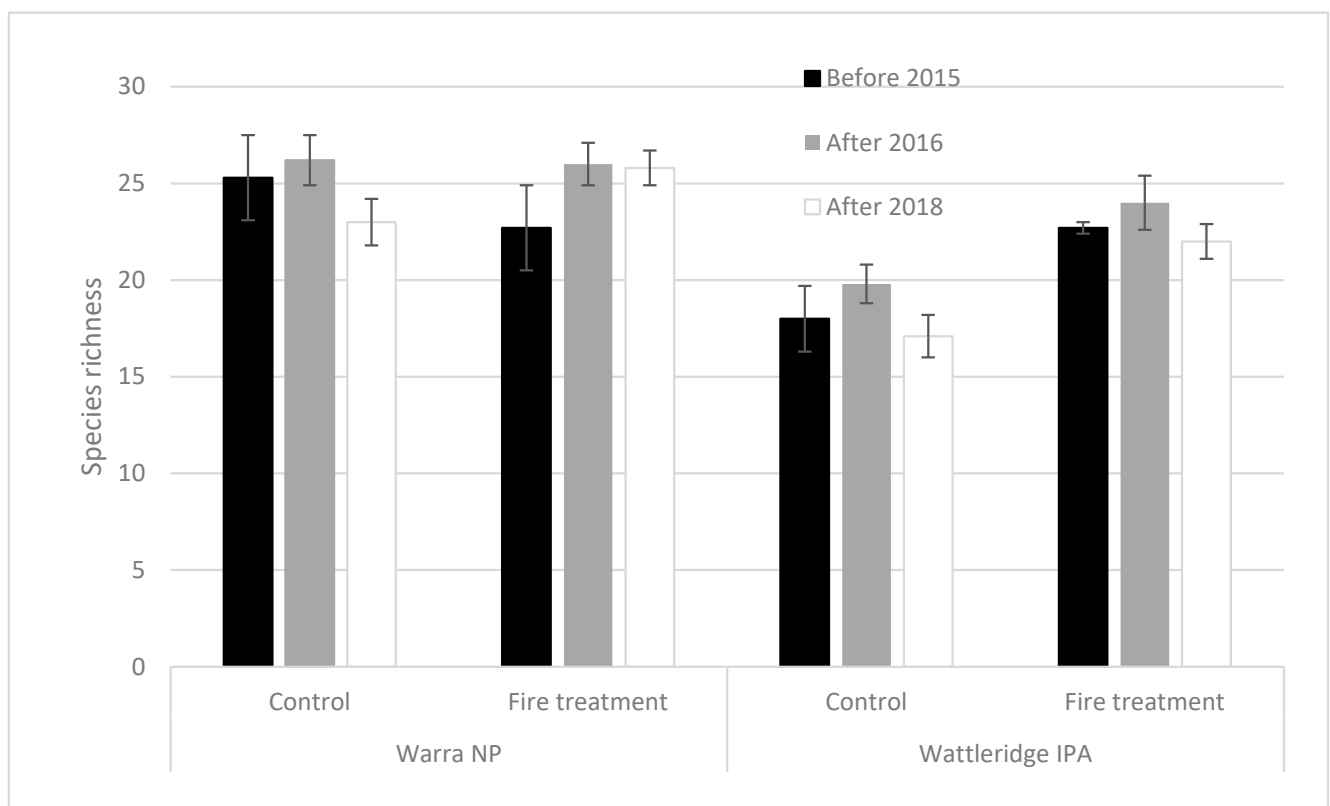


Figure 3. Mean herbaceous species richness (± 1 SE) at Warra NP and Wattleridge IPA in control and fire treatment (impact) plots before (2015) and after (2016 and 2018) treatment.

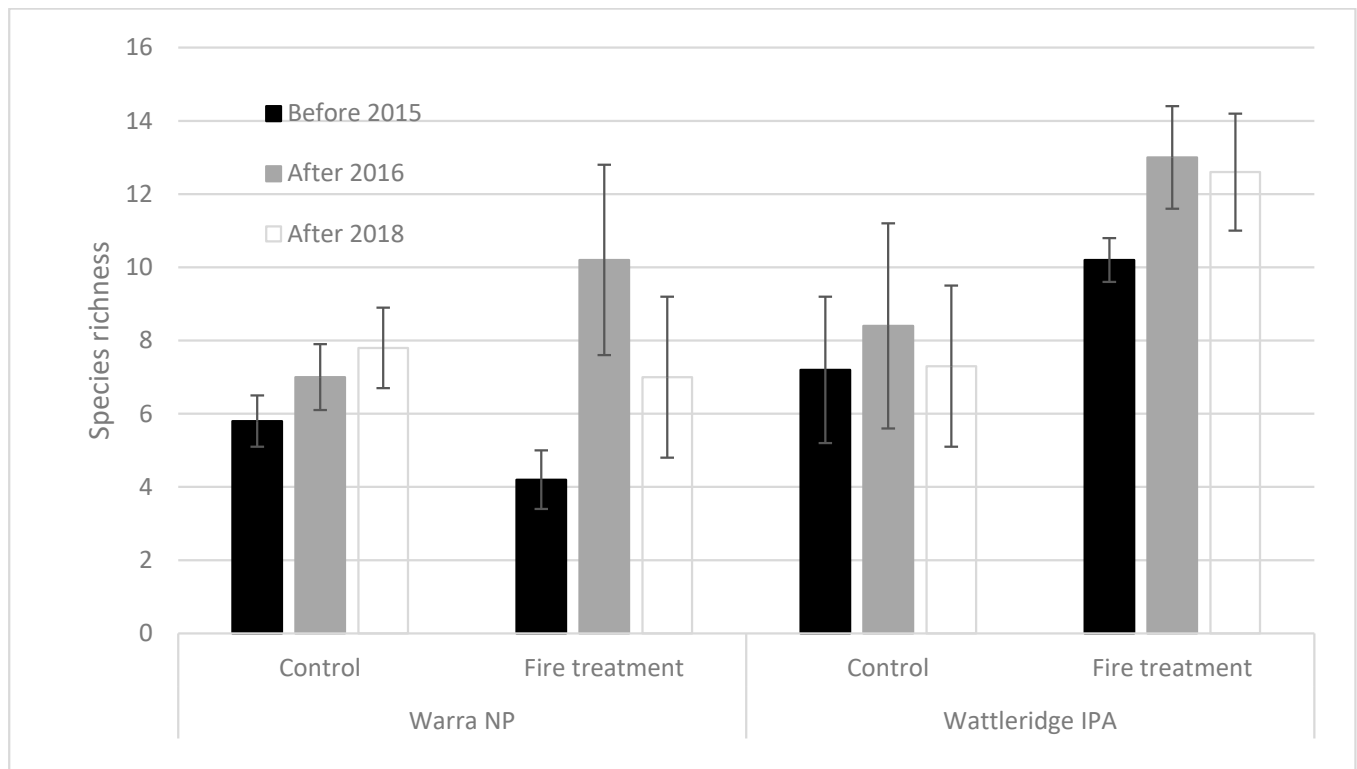


Figure 4. Mean species richness (± 1 SE) of shrubs and juvenile trees at Warra NP and Wattleridge IPA in control and fire treatment (impact) plots before (2015) and after (2016 and 2018) treatment.

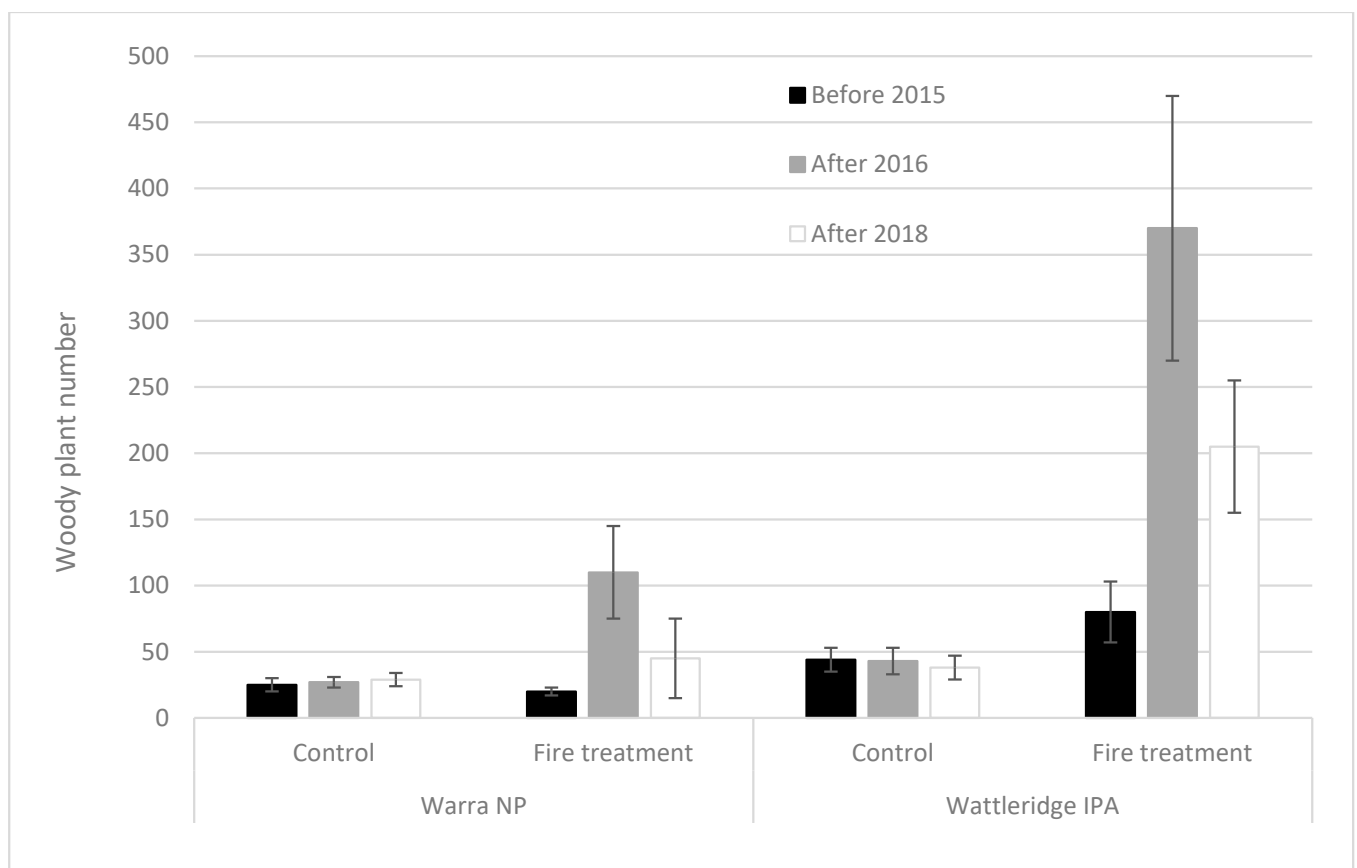


Figure 5. Mean shrub and juvenile tree abundance (± 1 SE) at Warra NP and Wattleridge IPA in control and fire treatment (impact) plots before (2015) and after (2016 and 2018) treatment.

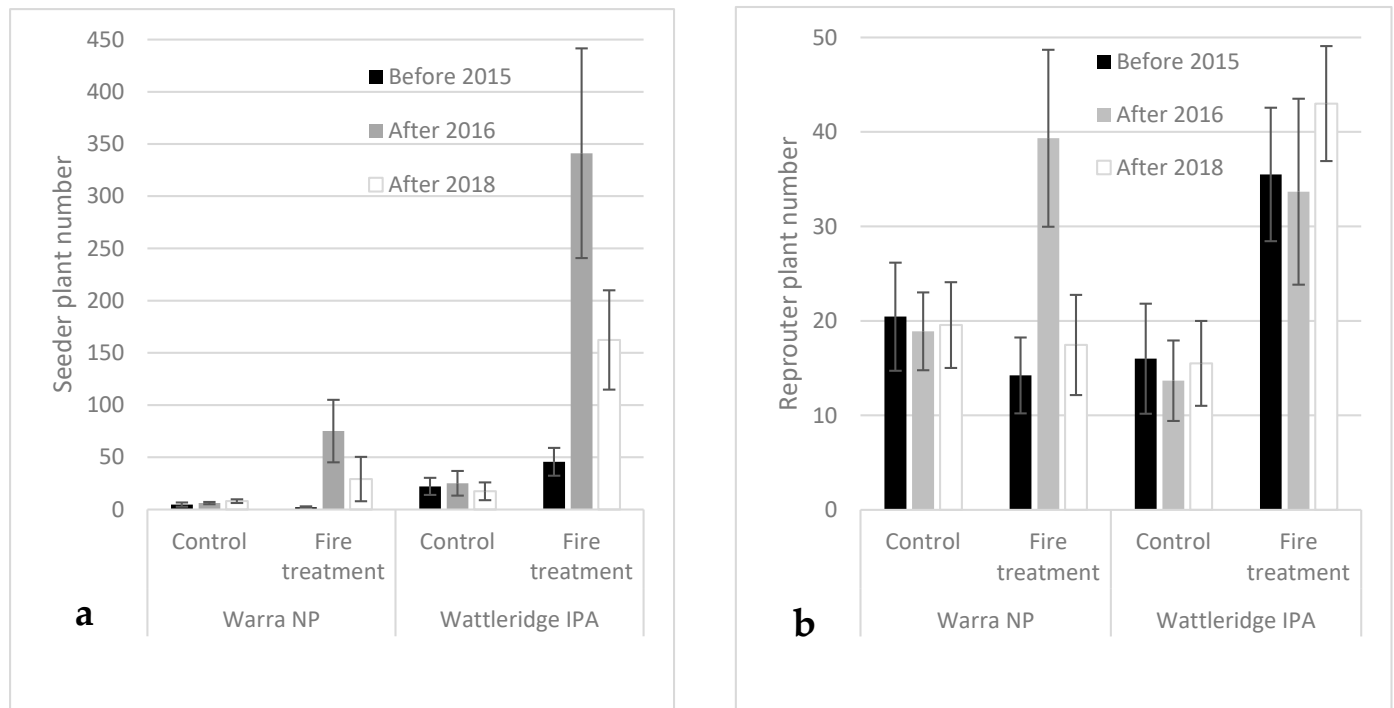


Figure 6. Mean seeder (a) and resprouter (b) plot abundance (± 1 SE) at Warra NP and Wattleridge IPA in control and fire treatment (impact) plots before (2015) and after (2016 and 2018) treatment.

4. Discussion

Our study breaks new ground in fire ecology by undertaking quantitative analysis of cultural burning impacts on dry sclerophyll forests. We aimed to explore whether the benefits of cultural burning, as described by Indigenous fire practitioners, were supported by the findings of western science. Here, we first discuss the ecological results of the study and then link them to the potential benefits of cultural burning. We also outline the opportunities to develop meaningful cross-cultural knowledge further.

Low-severity cultural burning and moderate-severity hazard reduction burning did not have a significant impact on full-floristic composition (cover) or species richness of the vegetation overall or on the herbaceous layer in the 1–3 years after fire. Similarly, other studies of the ecological effects of light to moderate-severity prescribed burning in dry sclerophyll forests in southeast Australia have found that plant cover, abundance, and species richness were not significantly affected in the long term [75–77].

Only the higher severity hazard reduction burn had a significant impact on shrub and juvenile tree cover and composition. Species richness of shrub and juvenile trees was greater at Wattleridge than Warra NP. Shrub and juvenile tree abundance increased at both locations due to fire, more so at Wattleridge IPA where it remained elevated three years after fire. This was largely driven by a few species that were more common at Wattleridge IPA (e.g., *Hovea purpurea* and *Grevillea scortechinii* subsp. *sarmentosa*), probably due to a higher quantity of soil-stored seed due to the comparatively long unburnt status of the vegetation at Wattleridge (i.e., >30 years, Table 1). In contrast, shrub and tree species at Warra may have had depleted seed reserves due to the greater frequency of recent fire [32,75], resulting in a lower abundance of shrubs and juvenile trees. This is supported by Kitchin's [53] finding that frequent fire and short inter-fire intervals were associated with reduced shrub species and seed bank richness and abundance at nearby Guy Fawkes River National Park in northern NSW.

Other studies have found that time since fire affects species composition and plant functional type [78,79]. Chick et al. [80] found a positive relationship between seed bank

richness and time since fire in heathy woodland ecosystems in Victoria, suggesting that seed bank diversity accumulates over time. However, in their study, spatial and environmental variability had a greater influence on seedbank composition than time since fire. We also found minor botanical differences between Warra and Wattleridge, despite their proximity, and seasonal effects between years unrelated to fire treatment and probably related to variable annual rainfall. Under certain circumstances, these confounding effects may have a larger effect on variables such as species richness than fire.

On the New England Tablelands, Clarke et al. [81] found partial support for a fire frequency disturbance model which predicted that ‘resprouting will be more frequent in landscapes with higher probabilities of frequent fire’. They [43] found that 63% of woody plants in New England dry sclerophyll forests were resprouters whereas our study found that 33% were resprouters at Wattleridge IPA and 45% at Warra NP. At a broader scale, the hypothesis that resprouting is positively correlated with fire frequency or intensity has been supported by some studies [82–85] but not others [44]. Clarke et al. [43] suggested that resprouting is a labile trait that varies within genera and even within species, due to environment or genotype.

The long unburnt fire regime may have made the Wattleridge vegetation more responsive to fire than the more frequently burnt vegetation at Warra, such that the mosaic cultural burn had a greater impact on woody species. When comparing cultural burning to other fire treatments, results need to be considered carefully within the specific context of each treatment. The response of a frequently burnt ecosystem may be subdued despite an extensive, more intense fire, whereas a patchy, low-intensity fire in a long unburnt ecosystem may have a greater impact, given the potential for the seed bank of seeder species to have increased over time. Fire history and regime are important considerations when interpreting the outcomes of fires and in fire management planning.

The hazard reduction burn at Warra was within the lower limit of the fire interval threshold, which is based on vital attributes of plant species. In comparison to Wattleridge, which was long unburnt, Warra had a reduced shrub diversity and abundance, a subdued seeder fire response, and a skewed ratio towards resprouters. This is possibly a result of too frequent burning, despite the fire management at Warra being within threshold. This suggests the Ecologically Sustainable Fire Management [56] thresholds need reviewing, so as not to lose shrub—and particularly seeder—diversity over time. Despite some differences in fire regime and vegetation communities across the locations, the precautionary principle applies—and in this case, fire management guidelines for Warra should be reviewed.

In December 2019, the extensive (9091-ha) high-severity Crown Mountain wildfire burnt all of the control and impact BACI plots at Wattleridge IPA and Warra NP. This provides an opportunity to compare the impact of wildfire on the variables addressed in this paper, and to compare the severity and impacts of wildfire in the long unburnt control plots with high fuel loads, compared to the 2015 culturally burnt plots with lower fuel loads. McKemey et al. [32] studied the impact of cultural burning on the threatened Backwater grevillea, *Grevillea scortechinii* subsp. *sarmentosa*, at Wattleridge IPA, and found that cultural burning had a lesser impact on mature grevillea survival than the wildfire. The cultural burn created a multi-age population in culturally burnt sites, with mature surviving shrubs contributing to the seed bank and seedlings providing the next generation of grevilleas. It would be worthwhile to extend the research to other species in this study.

Our research addresses the knowledge gap relating to the quantitative ecological assessment of Indigenous cultural burning in southeast Australia. Cultural burning has been asserted to provide ecological and bushfire management benefits, as described in the Introduction. This study, although a single case-study, provides support for some of these claims. For example, we found that both cultural burning and hazard reduction burning decreased

fuel loads. Although the Crown Mountain wildfire subsequently burnt both Warra NP and Wattleridge IPA in 2019, it is unlikely that any prescribed burning program could have prevented the unprecedented, extreme ‘Black Summer’ wildfires of 2019–2020 [32,86–90]. Our study found that the low-severity cultural burn did not significantly impact full-floristic or herbaceous species richness or cover but stimulated the regeneration of seeder shrubs and trees. The cultural burn in 2015 could potentially have depleted the seed bank and the regenerative reserves of resprouters, which would then have been further diminished by the wildfire in 2019. Frequent fire can result in the disruption of life-cycle processes in plants and loss of vegetation structure and composition [54], so future fire planning by the Banbai rangers should take into account these potential impacts on population viability of the constituent species in the Wattleridge vegetation. The ecosystems and biodiversity values at Wattleridge were in good condition prior to cultural burning [64,70], so restoration of ecosystems could not be demonstrated in this study. Banbai rangers undertook cultural burning through a cultural management framework, which allowed Indigenous custodians to conserve biodiversity using culturally driven, holistic management that was locally focused and supported by cross-cultural knowledge [32]. Banbai cultural fire management afforded an array of cultural benefits, as discussed in McKemey et al. [32,37,38]. In terms of ecological and bushfire management outcomes, this study provides evidence to support claims that cultural burning decreases fuel loads, stimulates regeneration of seeder shrubs and trees, and provides management at a local place-based scale, while not significantly impacting full floristics or the herbaceous vegetation layer.

The scale, size, timeframe, and fire histories related to this study were insufficient to answer ecological questions of long-term population dynamics and post-fire survival of plant species. Due to the complexities of prescribed fire management, fire treatments were applied at different sites, which limits the strength of comparisons between cultural burning and hazard reduction. This constrains interpretation of the differences between cultural and non-cultural burning. Nonetheless, this study demonstrated no negative impact of cultural burning on vegetation communities. It also assisted cross-cultural knowledge development by working together to apply western scientific inquiry to Indigenous cultural burning. To achieve meaningful cross-cultural outcomes, such studies should continue into the long term and focus on the continued application of two-way science [91–93] over relevant time periods (i.e., decades). This approach is facilitated by the place-based management of Indigenous communities, given their inter-generational desire to look after their Country. Even a decades-long study is a snapshot in the context of thousands of generations of land management [15,94]. Regardless, studies such as ours are part of the cross-cultural science of applied fire ecology, and part of the learning process contributing to the adaptive management of Indigenous protected areas and national parks. On the basis of our results, we recommend that cultural burning be a key management tool in Indigenous protected areas—and other land tenures—and that it be monitored and managed adaptively through two-way science with methods such as ‘reading Country’ [13] and western ecological survey techniques [95], in order to lead to culturally and ecologically beneficial fire regimes. This is a crucial component of our resilience for the Pyrocene [96,97] and an important step in the decolonization of science and land management [17,98]. As co-author, Aboriginal Elder Lesley Patterson states, ‘Cultural burning is a part of our DNA—we must look after Country and make sure we do the right thing by Country. We should practice cultural burning on all land tenures to make sure that non-Indigenous people know there are good benefits from cultural burning. That’s what our Country was born out of, and our old people [ancestors] knew what they were doing. We’ve got to be the leaders and show the benefits for Australia. It’s the circle of life, and we need the fire to rejuvenate life.’

5. Conclusions

Similar to other studies, we found that low- and moderate-severity burns did not have significant impacts on vegetation composition and cover, including the herbaceous layer, over a three-year period in dry sclerophyll forests on the New England Tablelands in northern NSW. However, frequent burning in Warra National Park may be leading to reduced shrub diversity and abundance, a subdued seeder fire response, and a skewed ratio towards resprouters. We found the cultural burn stimulated a mass germination of woody seeders in vegetation that had not been burnt for at least 30 years, emphasizing the importance of fire history when comparing the impacts of various fire treatments. Through this case study, we provide empirical evidence that supports claims that Indigenous cultural burning provides ecological and bushfire management benefits.

Supplementary Materials: The following supporting information can be downloaded at: <https://www.mdpi.com/article/10.3390/fire8090367/s1>, Supplementary Material S1—PERMANOVA and ANOVA outputs; Supplementary Material S2—Table S1: Fire responses of individual shrub and tree taxa.

Author Contributions: Conceptualization, N.C.H.R., J.T.H. and M.M.; methodology, N.C.H.R., J.T.H. and M.M.; formal analysis, N.C.H.R., J.T.H. and M.M.; investigation, N.C.H.R., J.T.H., M.M., M.P. and I.S.; resources, N.C.H.R. and M.M.; data curation, M.M. and J.T.H.; writing—original draft preparation, M.M.; writing—review and editing, N.C.H.R., J.T.H. and M.M.; supervision, N.C.H.R., J.T.H. and M.P.; project administration, N.C.H.R.; funding acquisition, N.C.H.R. and M.M. All authors have read and agreed to the published version of the manuscript.

Funding: This study was funded by University of New England, Firesticks Project of the Nature Conservation Council NSW, Northern Tablelands Local Land Services through the National Landcare Program Australia, Rural Fire Service Association, and Rural Fire Service NSW.

Institutional Review Board Statement: The study was conducted under the University of New England Human Ethics approval HE14–182 and 19–068. A scientific licence was granted by the NSW Government to undertake research in Warra NP and Wattleridge IPA, licence number SL101661.

Informed Consent Statement: Informed consent was obtained from all subjects involved in the study.

Data Availability Statement: The data presented in this study are available on request from the corresponding author.

Acknowledgments: We would like to acknowledge the Banbai Rangers and all Indigenous people, past and present, who have cared for and shared their knowledge of Country and culture. Thanks to Emilie Ens, Oliver Costello, and Malcolm Ridges for PhD co-supervision, Catherine MacGregor for assistance with map preparation, and to Koen Dijkstra, Peter Croft, and Vanessa Hunter. We acknowledge NPWS and Banbai Land Enterprises for allowing us to undertake research on their land.

Conflicts of Interest: The authors declare no conflicts of interest.

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