Simulating the effectiveness of prescribed burning at altering wildfire behaviour in Tasmania, Australia

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\textbf{Abstract.} Prescribed burning is a widely accepted wildfire hazard reduction technique; however, knowledge of its effectiveness remains limited. To address this, we employ simulations of a widely used fire behaviour model across the ecologically diverse Australian island state of Tasmania. We simulate three broad scenarios: (1) no fuel treatment, (2) a maximal treatment, with the most possible prescribed burning within ecological constraints, and (3) 12 hypothetically more implementable state-wide prescribed-burning plans. In all simulations, we standardised fire-weather inputs to represent regionally typical dangerous fire-weather conditions. Statistical modelling showed that an unrealistically large maximal treatment scenario could reduce fire intensity in three flammable vegetation types, and reduce fire probability in almost every vegetation type. However, leverage analysis of the 12 more-realistic implementable plans indicated that such prescribed burning would have only a minimal effect, if any, on fire extent and that none of these prescribed-burning plans substantially reduced fire intensity. The study highlights that prescribed burning can theoretically mitigate wildfire, but that an unrealistically large area would need to be treated to affect fire behaviour across the island. Rather, optimisation of prescribed burning requires careful landscape design at the local scale. Such designs should be based on improved fire behaviour modelling, empirical measurement of fuels and analysis of actual wildfires.

\textbf{Additional keywords:} dry eucalypt forests, eucalypt forests, fire behaviour model, fire management, fire weather, fuel reduction, fuel treatment, leverage, sedgelands, temperate rainforests, wet eucalypt forests.

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\textbf{Introduction}

There is widespread acceptance that prescribed burning is an essential component of effective wildfire management (Fernandes and Botelho 2003; Moritz et al. 2014). Indeed, a consistent outcome of every major inquiry into bushfire disasters in Australia has been a recommendation for an increase in fuel treatment through prescribed burning (Ellis et al. 2004; Teague et al. 2010; Attiwill and Adams 2013). However, prescribed burning is an imperfect form of wildfire hazard management, as its benefits are dependent on the ecological and geographical context (Moritz et al. 2014). Prescribed fires carry the risk of escape, and are unsuitable in many vegetation types because the fires cannot be controlled safely (Bradstock et al. 1998; Bradstock et al. 2005; Lindenmayer and Burgman 2005; Penman et al. 2011). Additionally, prescribed burns carry significant side effects, such as ecological harms, both through degrading fauna and habitat (Catling et al. 2001; Andersen et al. 2005) and disadvantageing some plant species that require long fire-free intervals to complete their life cycle (Enright et al. 2015). Another side effect has been shown to be negative consequences to human health from smoke (Williamson et al. 2016a). Because of these constraints, there is considerable debate, both scientific and political, about the utility of prescribed burning as a management strategy in mitigating fire activity (Morrison et al. 1996; Fernandes and Botelho 2003; Penman et al. 2011; Stephens et al. 2016).

Prescribed burning affects the behaviour of subsequent wildfires by reducing fuel loads in flammable areas across a landscape. By increasing the amount of recently burnt areas, managers can increase the encounter rate of unplanned fires with areas containing low fuel loads, hence mitigating fire spread across the landscape (Finney 2001; Fernandes and Botelho 2003; Agee and Skinner 2005; Price 2012). For instance, analysis of landscape fire patterns in Australian dry eucalypt (Eucalyptus) forests has shown that, depending on the region, between three and ten areal units of prescribed burning can reduce wildfire extent by one areal unit (Boer et al. 2009; Price et al. 2015a). However, this ratio, known as ‘leverage’ (Loehle 2004), has been shown to be highly variable among different vegetation types globally (Price et al. 2015b).

The ability of prescribed burning to reduce fireline intensities is an equally important reason for its application. Under moderate fire-weather conditions, intensities of \( \sim 4000 \text{ kW m}^{-1} \) are considered the upper limit for direct attack suppression. Above \( 10000 \text{ kW m}^{-1} \), fire behaviour is considered extreme, and it is recommended that all firefighting activities, including at the
flanks and back of the fire, be ceased (Hirsch and Martell 1996; Fernandes and Botelho 2003). Under more extreme weather conditions, eucalypt forests in south-east Australia are reported to be capable of sustaining maximum fireline intensities of $\sim 100,000$ kW m$^{-2}$ (Gill and Moore 1990; McCarthy et al. 1999). Such fire-weather conditions result in fuel loads and disturbance history being less important in driving fire intensity in this vegetation type, placing an upper bound on the effectiveness of prescribed burning (Bradstock 2010; Price and Bradstock 2011; Price and Bradstock 2012; Bowman et al. 2016). However, current understanding the effect of prescribed burning on fire intensity in dangerous fire weather needs to be extended to other vegetation types.

There are substantial practical difficulties in conducting landscape-level fire experiments (e.g. Gould et al. 2008), so consequently much of the evidence-base that supports the effectiveness of prescribed burning in mitigating wildfire damage rests on case studies and expert opinion (e.g. Underwood et al. 1985; Grant and Wouters 1993; Fernandes and Botelho 2003), retrospective landscape-level analyses of wildfire severity (e.g. Hammill and Bradstock 2006; Bradstock 2010; Price and Bradstock 2012) and computer simulations (e.g. Bradstock et al. 2012; Penman et al. 2013). Simulation studies are of particular importance because they enable exploration of the effects of a wide range of fire-weather and fuel-load scenarios for different vegetation types (e.g. Beverly et al. 2009; Bradstock et al. 2012). Simulation studies for south-east Australian dry eucalypt forests have reinforced a key limitation of prescribed burning by showing that, under extreme fire weather, the reduction in fuel loads is of negligible importance (Cary et al. 2009; Penman et al. 2013; Price et al. 2016). In contrast, in North American (Beverly et al. 2009) and Mediterranean (Carmel et al. 2009) landscapes, simulations have been used to show that, under dangerous fire-weather conditions, vegetation type (and hence fuel type) is the primary determinant of intensity. However, similar studies explicitly investigating the effect of vegetation and fuel type on fire behaviour are currently lacking in Australia.

Understanding the comparative effectiveness of prescribed burning in different vegetation types under dangerous fire weather is of prime importance in the ecologically diverse, temperate, continental island state of Tasmania, 200 km south of Victoria, Australia. The island is $\sim 68,000$ km$^2$, (with an additional 12,000 km$^2$ in off shore islands) and supports diverse vegetation which interplay of a steep precipitation gradient from the west to east, rugged terrain and varied geology (Reid et al. 1999; Colhoun and Shimeld 2012). The perhumid west of the island is characterised by a complex mosaic of flammable and fire-sensitive vegetation types that include highly flammable treeless sedgelands (locally known as button-grass moorlands) and scrubland, tall wet eucalypt forests that can support some of the highest fire intensities on Earth, and Gondwanan rainforests, which are fire sensitive (Hill 1982; McCarthy et al. 1999; Marsden-Smedley et al. 2001). In the drier east of the island there are large swaths of dry eucalypt forests, interspersed with smaller areas of flammable native grassland, as well as the occasional tract of wet forest, rainforest or coastal scrub. It is in these eastern areas that the state’s agricultural enterprises and settlements are embedded (Fig. 1a).

The state’s 11 broad vegetation types can be generally divided into three groups based on the geographic extent to which fire managers consider them to be safely treatable through prescribed burning. Three vegetation types are extensively treatable (i.e. $>80\%$ of the vegetation’s area is available for prescribed burning), these are dry eucalypt forest, sedgeland and native grassland. Three vegetation types are only partially treatable (i.e. $<45\%$ of the vegetation’s area is available for prescribed burning), these are non-eucalypt forest, native scrublands and non-native vegetation. Five are completely untreatable with prescribed burning (wet eucalypt forest, rainforest, highland vegetation, wetlands and other vegetation types) (Table 1).

Compared with mainland south-eastern Australia, the island experiences much less frequent dangerous fire weather (Fox-Hughes 2008; Williamson et al. 2016b), yet since European settlement in 1803, there have been a number significant and costly fire disasters in Tasmania. These include five poorly documented major fire events between 1898 and 1960 (Luke and McArthur 1978), the notorious 1967 fire that nearly destroyed the state capital, Hobart (Solomon and Dell 1967), and, more recently, the Dunalley fire disaster in summer of 2012–13 (Henley 2013; Hyde 2013) and the wilderness fires in 2015–16 (Commonwealth of Australia 2016). Climate change projections indicate dangerous fire weather will increase through this century across the island (Fox-Hughes et al. 2014). As a consequence of the increased threat of destructive wildfires, the Tasmanian government has made a commitment to substantially increase the area of both public and private land treated with prescribed burning to mitigate wildfire risk across the whole island (Richards et al. 2014). However, current understanding of the benefits and limitations of this intervention remains limited. Particularly, the trade-offs of prescribed burning among different vegetation types need to be better understood. Resolving such a complex question demands simulation studies. In this study we use simulations of a fire behaviour model to address three questions:

1. Under conditions characteristic of a dangerous fire-weather day that would occur three to four times per fire season, what fire intensities are Tasmania’s wildlands capable of sustaining, and how do they vary geographically?
2. Under a maximal, but unrealistic, prescribed-fire regime, one that includes the most possible prescribed burning on all land considered treatable ($\sim 31\%$ of Tasmania, based on vegetation type, land use type and land tenure), what is the capacity of prescribed burning to reduce the intensity, and extent, of wildfire under bad fire-weather conditions?
3. How effective are 12 hypothetical but more implementable prescribed-burning plans (which cumulatively treat between 1 and 25% of the total area treated under the maximal scenario, and have been designed by fire management agencies) at reducing the intensity and extent of unplanned fires under bad fire weather?

In our simulations we use the Phoenix RapidFire model (hereafter Phoenix) (Tolhurst et al. 2008): a spatially and temporally explicit fire behaviour model that has become the standard fire behaviour model for south-east Australian fire agencies. We simulated 11,059 fires, standardising weather
conditions to be those of a typically dangerous fire-weather day, defined at the 99th–99.5th percentile of the Forest Fire Danger Index (FFDI) in each of 45 meteorological regions across Tasmania (Richards et al. 2014; Fig. 1c). We varied fuel loads according to three scenarios described above (see Table S1, available as Supplementary material to this paper). We analysed geographic trends in fire intensity qualitatively, through mapping, and quantitatively analysed the simulation results through statistical modelling. The latter exercise provided an insight into the importance of fire weather, terrain (slope and aspect), vegetation type and fuel loading in affecting the outputs of the Phoenix model. We also used the metric ‘leverage’, which quantifies the spatial effectiveness of fuel treatments at a regional scale (Loehle 2004), to understand the effect of fuel reduction on fire behaviour in Tasmania apparent in our analysis of implementable prescribed-burning plans. In addition, we calculated a
novel variant of the leverage metric to analyse the efficiency of these implementable plans in reducing fireline intensity.

Methods

Geographic framework

The simulation study was organised in a Geographic Information System (GIS) environment using a lattice of 11 059 ignition points spaced 2.5 km apart, covering the entire mainland of Tasmania and outlying islands. Our decision to use a uniform grid of ignitions was a pragmatic approach based on computing constraints that obviated the use of more complicated, but realistic, designs. Biogeographic and climactic data were organised into 25 × 25-m cells. Each cell was assigned 1 of 11 vegetation types, based on TASVEG, ver. 3.0 (Tasmanian Vegetation Monitoring and Mapping Program, Department of Primary Industries, Parks, Water and Environment, Hobart, Tas., Australia) – abbreviated names used in this paper are also given. Vegetation types considered treatable by prescribed burning are in bold. The proportion (%) of the Tasmanian landscape covered by each vegetation type, and the proportion of vegetation that is treated with prescribed burning under the maximal scenario (see Fig. 1b, available as Supplementary material to this paper) is indicated.

<table>
<thead>
<tr>
<th>Vegetation type (TASVEG classification)</th>
<th>Abbreviated name</th>
<th>Total area (km²)</th>
<th>Percentage Tasmanian landscape</th>
<th>Percentage treated area</th>
</tr>
</thead>
<tbody>
<tr>
<td>Agricultural, urban and exotic vegetation</td>
<td>Non-native vegetation</td>
<td>16200</td>
<td>23.6</td>
<td>5.08</td>
</tr>
<tr>
<td>Dry eucalypt forest and woodland</td>
<td>Dry eucalypt forest</td>
<td>15900</td>
<td>23.16</td>
<td>92.04</td>
</tr>
<tr>
<td>Wet eucalypt forest and woodland</td>
<td>Wet eucalypt forest</td>
<td>10900</td>
<td>15.88</td>
<td>0</td>
</tr>
<tr>
<td>Rainforest and related scrub</td>
<td>Rainforest</td>
<td>7160</td>
<td>10.43</td>
<td>0</td>
</tr>
<tr>
<td>Moorland, sedgeland, rushland and peatland</td>
<td>Sedgeland</td>
<td>5990</td>
<td>8.73</td>
<td>96.06</td>
</tr>
<tr>
<td>Scrub, heathland and coastal complexes</td>
<td>Native scrublands</td>
<td>5130</td>
<td>7.47</td>
<td>43.53</td>
</tr>
<tr>
<td>Other natural environments</td>
<td>Other environments</td>
<td>2530</td>
<td>3.69</td>
<td>0</td>
</tr>
<tr>
<td>Non eucalypt forest and woodland</td>
<td>Non-eucalypt forest</td>
<td>2140</td>
<td>3.12</td>
<td>19.83</td>
</tr>
<tr>
<td>Native grassland</td>
<td>Native grassland</td>
<td>1370</td>
<td>2</td>
<td>80.66</td>
</tr>
<tr>
<td>Highland and treeless vegetation</td>
<td>Highland vegetation</td>
<td>1080</td>
<td>1.57</td>
<td>0</td>
</tr>
<tr>
<td>Saltmarsh and wetland</td>
<td>Wetlands</td>
<td>240</td>
<td>0.35</td>
<td>0</td>
</tr>
</tbody>
</table>

Fire behaviour simulations

This study, which was part of a larger fire behaviour model simulation study commissioned by the Tasmania State Fire Management Council (Richards et al. 2014), consisted of 62 repeated implementations of the grid of 11 059 ignition points mentioned above, with each implementation occurring under a different prescribed burning scenario. For each implementation of this ignition grid, hereafter referred to as a series of simulations, the Phoenix RapidFire (Tolhurst et al. 2008) fire-behaviour model was used to simulate an ignition at each point. Based on assigned values for topographic variables (slope and aspect), percentile FFDI day. Even though the variables were selected independently for each hour, there is an intrinsic correlation between hourly observations, as all the observations were from the same 99th to 99.5th percentile FFDI days. The end result was a single synthetic day for each of the 45 climate zones (Richards et al. 2014) for use in the fire behaviour modelling.

<table>
<thead>
<tr>
<th>Meteorological variable</th>
<th>Minimum</th>
<th>Mean</th>
<th>Maximum</th>
<th>Percentile (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Temperature (°C)</td>
<td>21</td>
<td>30.1</td>
<td>35</td>
<td>85</td>
</tr>
<tr>
<td>Drought factor</td>
<td>6.8</td>
<td>8.5</td>
<td>10</td>
<td>Mean</td>
</tr>
<tr>
<td>Wind speed (km h⁻¹)</td>
<td>15.7</td>
<td>25.6</td>
<td>39.2</td>
<td>Mean</td>
</tr>
<tr>
<td>Humidity (%)</td>
<td>15.3</td>
<td>26.4</td>
<td>49.2</td>
<td>15</td>
</tr>
<tr>
<td>Cloud cover (%)</td>
<td>0</td>
<td>2.1</td>
<td>30</td>
<td>25</td>
</tr>
</tbody>
</table>
fire spread was independently simulated for each ignition across the grid of 25 × 25-m cells between 1300 and 1900 h. The fire was assumed to be under the influence of the synthetic weather day based on the ignition point’s associated climate zone, regardless of whether the fire burned into an adjacent climate zone. Fuel load was estimated based on fire history using fuel accumulation curves specific to each vegetation type. These accumulation curves were developed using published and unpublished empirical field studies (Richards et al. 2014). The fuel loads were derived from these curves as a function of time since the previous fire based on fire maps up to 2013 (‘ambient fuel loading’), then varied as described below. Model outputs were averaged up to a 200 × 200-m grid.

Fuel loads were varied according to three broad scenarios: (1) a null treatment, (2) a maximal treatment, and (3) 12 implementable prescribed-burning plans, as designed by fire managers (Richards et al., 2014), spaced over 5 years (i.e. 60 series of simulations) (see Table S1). Under the null treatment scenario, a series of fire-behaviour simulations were performed on a landscape with the ambient 2013 fuel load. Under the maximal treatment scenario, the ambient fuel load was modified by setting fuel age to zero on all land considered to be treatable by prescribed burning (Fig. 1b) and then a series of simulations were performed. A parcel of land was considered treatable if the vegetation type could generally tolerate prescribed burning (based on Kitchener and Harris 2013 and Pyrke and Marsden-Smedley 2005) and if the land use type and tenure allowed for prescribed burning (Richards et al., 2014).

Under the 12 implementable plans, different areas of the landscape were selected for treatment using several selection techniques. These techniques were designed by the Tasmanian State Fire Management Council using ‘science-based risk management principles’ with the goal of being ‘realistic and measurable’ (Richards et al. 2014). Each of the 12 plans simulated treatment of a given area of the landscape each year between 2014 and 2018, and a series of simulations were performed after each annual treatment. Treatment was simulated by setting the fuel load to zero and then allowing fuel to re-accumulate in future years according to the accumulation curves. A key feature of these plans is that several of them were based on a tenure-blind approach – treating areas of public and private land indiscriminately. It is important to note that these plans were designed to compare the relative effectiveness of different broad fuel-management strategies in the Tasmanian context and do not represent actual management plans to be implemented. For more details on each of the treatments, see Table S1 and Richards et al. (2014).

For each series of simulations performed under each fuel-management scenario, we recorded the maximum intensity of all fires to pass through each 200 × 200-m cell to consider how prescribed fire mitigates the upper extremes of fire intensity. Maps were then produced showing the geographic pattern of maximum fire intensity across the state under the null and maximal treatment scenarios.

**Analyses of fire intensity**

We mapped the variables that underpinned our fire behaviour model simulations: vegetation type, the extent of treatable vegetation, the Bureau of Meteorology’s climate zones and their associated 99th–99.5th percentile FFDI, and digital elevation as a representation of terrain (Fig. 1). We compared these maps qualitatively to look for similar spatial patterns in fire intensity so that we could visually assess which factors most were most likely to be driving maximum fire intensity in the simulations. We then used generalised linear models (GLMs) to investigate the relative importance of the various Phoenix Rapidfire input variables in driving fire intensity and probability of fire presence. Owing to computing constraints, we analysed fire intensity for a subset of 10 297 output cells selected using the random points function in ArcGIS geospatial software (ESRI Inc., Redlands, CA, USA, www.esri.com). We removed any points that were overlying lakes, reservoirs or watercourses. To smoothe the slope and aspect data to have the same resolution as the Phoenix output grid we used a two-step cubic then bilinear resample in ArcGIS. We then performed a cosine transformation on aspect to ensure circular continuity, but as a result, the variable only contained north–south aspect information.

To determine the relative influence of the aforementioned variables on Phoenix model predictions of fire presence and intensity we used a two-stage statistical modelling analysis. First, to represent fire extent, we modelled the probability of fire presence in a given cell using a binomial GLM with a logit-link function. Then, we modelled the maximum fire intensity for each cell in which fire had occurred using a gamma GLM with a logarithmic-link function. In both cases, the response variable was modelled as a function of general vegetation type, 99th–99.5th percentile FFDI (a measure of regional climate), topography and the treatment scenario. A summary of response and explanatory variables and their abbreviations is given in Table 3. The full models (defined as the models with the most support) were:

\[
P(\text{FIRE}) \sim \text{TRT} + \text{VEG} + \text{TRT} \times \text{VEG} + \text{FFDI} + \text{SLP}
\]

(1)

\[
\text{INTENS} \sim \text{TRT} + \text{VEG} + \text{TRT} \times \text{VEG} + \text{FFDI} + \text{SLP} + \text{ASP}
\]

(2)

To determine the relative explanatory power of each variable, we compared the full models with a model for each explanatory variable containing only that explanatory variable. To identify the best predictors of fire intensity and presence within the Phoenix

<table>
<thead>
<tr>
<th>Variable</th>
<th>Description</th>
<th>Type</th>
</tr>
</thead>
<tbody>
<tr>
<td>INTENS</td>
<td>Fire intensity (kW m⁻¹)</td>
<td>Continuous</td>
</tr>
<tr>
<td>FIRE</td>
<td>Presence of fire</td>
<td>Binary</td>
</tr>
<tr>
<td>TRT</td>
<td>Treatment scenario (Null = 0, Maximal = 1)</td>
<td>Binary</td>
</tr>
<tr>
<td>VEG</td>
<td>General vegetation type</td>
<td>Categorical</td>
</tr>
<tr>
<td>FFDI</td>
<td>99th–99.5th percentile FFDI in associated climate zone</td>
<td>Continuous</td>
</tr>
<tr>
<td>SLP</td>
<td>Slope (degrees)</td>
<td>Continuous</td>
</tr>
<tr>
<td>ASP</td>
<td>Cosine-transformed north–south aspect</td>
<td>Continuous</td>
</tr>
</tbody>
</table>
model, we ranked the variables using difference in Akaike information criterion (ΔAIC) between each model and the full model. We also looked at deviance explained by each model. To investigate the utility of the more implementable prescribed-burning plans in different vegetation types, we used a leverage approach similar to that of Price et al. (2015a). Area-burnt leverage was defined as the reduction in area of unplanned fires associated with one unit of previous prescribed burning within each vegetation type. This was calculated as the negative slope of a linear model predicting the area of each vegetation type that burned in each series of simulations as a function the cumulative area treated in all previous years. We also introduce a new leverage metric, intensity leverage, which is calculated in the same manner as area-burnt leverage, except using models that predict intensity as a function of cumulative area treated. To calculate these intensities, we used a response variable for the linear models defined as the mean of maximum intensities experienced within burnt areas in each vegetation type. This metric gives a measure of how much 1 ha of treated area can reduce the aggregated worst-case scenario fire intensities across the island for each vegetation type on a bad fire-weather day. To calculate these leverage metrics, we used the results of simulations under the 12 implementable prescribed-burning plans spaced over 5 years, as outlined in Richards et al. (2014), with the outcomes of the 60 series of simulations each representing one data point (see Table S1). All modelling was performed in R (R Core Team, R Foundation for Statistical Computing, Vienna, Austria; http://www.R-project.org/).

Results
A striking feature of these fire behaviour simulations is the marked geographic variation in fire intensity across the island of Tasmania on a bad fire-weather day (99th to 99.5th percentile FFDI) under both the null and maximal fuel treatment scenarios (Fig. 2). This variation largely corresponds to vegetation type (Fig. 1a), and area treated (Fig. 1b). Based on simulated intensities (Fig. 3), Tasmania’s vegetation types can be divided into two broad groups: vegetation types with a substantial proportion of their area experiencing simulated intensities above the 10 000 kW m⁻¹ threshold, at which the cessation of firefighting is recommended (wet eucalypt forest, sedgelands, dry eucalypt forest and non-eucalypt forest), and vegetation types with approximately three-quarters or more of their area experiencing only fires below this threshold. Interestingly, in the three extensively treatable vegetation types (dry eucalypt forest, sedgelands and native grassland), the maximal treatment reduced intensities substantially, to the point where ~65–75% of each of the vegetation types’ areas experienced fires below the 4000 kW m⁻¹, at which suppression at the head of the fire is possible.

The GLMs and multi-model inference identified the relative importance of the various input variables in Phoenix RapidFire in affecting the likelihood of fire presence and fire intensity under the null and maximal treatment scenarios (Table 4), and provided further statistical support for the effect of the maximal fuel treatment. This treatment scenario differentially reduced

![Fig. 2. Geographic patterns of maximum simulated fire intensities in Tasmania under the (a) no fuel treatment scenario and (b) a maximal prescribed-burning scenario in which all treatable fuels were burnt state-wide (see Table 3).](image-url)
the probability of fire presence in all vegetation types, including in those not subjected to prescribed burning, except highland vegetation (Fig. 4a). By contrast, large, significant reductions in fire intensity (Fig. 4b) were only apparent in the three vegetation types that are considered by managers to be extensively treatable. However, in native scrublands, in which half of the area is considered treatable under the maximal treatment scenario, there was a small but potentially significant reduction (Fig. 4b). The statistical modelling revealed the predominant effect of vegetation in driving fire behaviour predictions (Table 4). It also identified slope and aspect as having important secondary effects on fire intensity, but not presence, a result consistent with the McArthur equations that underpin the Phoenix model (Noble et al. 1980). The effects of vegetation, slope and aspect on maximum fire intensity are most apparent for wet eucalypt forests in Tasmania’s south-east – where high biomass of flammable vegetation, which favours moist south facing slopes, burned with incredibly high intensities. These intensities were not diminished by a maximal prescribed-burning treatment (Figs 2, 4b).

In contrast to the maximal treatment scenario, the 12 implementable prescribed-burning plans, which treated much smaller areas of the state (cumulatively 1–25% of the area treated under the maximal scenario; see Table S1), had only a modest effect on fire extent and intensity. We analysed these treatment plans using the leverage metric and found that only grasslands, dry eucalypt forests, non-eucalypt forests and native scrublands exhibited area-burnt leverage, with values ranging from 0.2 to 0.28 (Fig. 5; Fig. S1, available as Supplementary material to this paper). These were much lower than estimated area-burnt

### Table 4. Comparison of the relative importance of fire behaviour model inputs in predicting the probability of fire presence and maximum fire intensity using generalised linear models (GLMs)

To assess relative importance, single variable models were compared with the model with the most support (first model listed). Models are ranked according to ΔAIC (the difference in Akaike Information Criteria between the best model and given model). Percentage deviance explained is relative to the null model, and indicates explanatory power. \( k \) is the number of parameters in the model. Variable abbreviations are defined in Table 3.

<table>
<thead>
<tr>
<th>Model</th>
<th>( k )</th>
<th>ΔAIC</th>
<th>Percentage deviance explained</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Probability of fire presence (P(FIRE))</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>TRT ( \times ) VEG + FFDI + SLP</td>
<td>24</td>
<td>0</td>
<td>23.1</td>
</tr>
<tr>
<td>TRT ( \times ) VEG</td>
<td>22</td>
<td>590</td>
<td>21.0</td>
</tr>
<tr>
<td>TRT + VEG</td>
<td>12</td>
<td>1590</td>
<td>17.4</td>
</tr>
<tr>
<td>TRT</td>
<td>2</td>
<td>3866</td>
<td>9.2</td>
</tr>
<tr>
<td>VEG</td>
<td>11</td>
<td>4504</td>
<td>7.0</td>
</tr>
<tr>
<td>FFDI</td>
<td>2</td>
<td>5987</td>
<td>1.7</td>
</tr>
<tr>
<td>SLP</td>
<td>2</td>
<td>6064</td>
<td>1.5</td>
</tr>
<tr>
<td>NULL</td>
<td>1</td>
<td>6470</td>
<td>NA</td>
</tr>
<tr>
<td><strong>Maximum fire intensity (INTENS)</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>TRT ( \times ) VEG + FFDI + SLP + ASP</td>
<td>25</td>
<td>0</td>
<td>38.9</td>
</tr>
<tr>
<td>TRT ( \times ) VEG</td>
<td>22</td>
<td>2342</td>
<td>27.2</td>
</tr>
<tr>
<td>TRT + VEG</td>
<td>12</td>
<td>3313</td>
<td>22.7</td>
</tr>
<tr>
<td>VEG</td>
<td>11</td>
<td>3318</td>
<td>21.6</td>
</tr>
<tr>
<td>SLP + ASP</td>
<td>3</td>
<td>3643</td>
<td>19.7</td>
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<tr>
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</table>

Fig. 3. Box and whisker plots of maximum simulated fire intensities under the maximal prescribed-burning scenario compared with the no fuel (null) treatment scenario for 10 of the vegetation types used in this study, ordered by the median intensity under the null treatment. The lines in the boxplots represent the first quartile, median, and third quartile; outliers were removed. The full range of simulated intensities can be found in the Table S2. Landscape cells with a maximum intensity of 0 kW m\(^{-1}\) (i.e. did not burn in the simulations) were not included in the plot. The vegetation type saltmarsh and wetlands was excluded, as there were insufficient values to be represented by a boxplot.
leverage values associated with the maximal treatment of ~0.5 to 1. Although it is not possible to directly calculate these latter leverage values, as the maximal treatment has only one data point, we estimated them using a ratio of reductions in area burnt (Fig. 4a) and treatable area (Table 1) under the maximal scenario. Additionally, only dry eucalypt forests and grasslands exhibited leverage on the fire intensities, with values of 0.0082 and 0.0097 kW m$^{-2}$/ha$^{-1}$. Interestingly, and in contrast to the maximal fuel treatment scenario, flammable sedgelands do not exhibit leverage in area burnt or intensity under these implementable plans.

**Discussion**

In this study, we used a fire behaviour model parameterised for typically dangerous fire weather (defined as the 99th to 99.5th percentile of the regional FFDI) and a grid of closely spaced ignitions across the whole of Tasmania to address three questions: (1) How does fire intensity vary among the state’s diverse vegetation types without any fuel treatment? (2) What is the upper bound of prescribed burning, namely, how can a maximal prescribed fire regime in dry eucalypt forest, sedgelands, native grassland, non-eucalypt forest, native scrublands and non-native...
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Fig. 5. Ordination of estimated area-burnt leverage and intensity leverage values (bars represent one standard error) for each of the Tasmanian vegetation types under the implementable prescribed-burning scenarios (12 treatments). Area burnt leverage is defined as the reduction in area of unplanned fire burning resulting from one unit area treated through prescribed burning. Intensity leverage is defined as the reduction in intensity in kilowatts per metre per hectare treated. Vegetation types not considered treatable cannot exhibit leverage and therefore are not included (Table 3). Non-native vegetation had a negligible amount of area treated, therefore the leverage estimates were not considered reliable and are not presented.

vegetation, all of which are considered by fire managers to be treatable with prescribed burning, reduce the intensity and extent of these simulated fires? (3) How effective might 12 hypothetical, more implementable prescribed-burning plans, as proposed by fire managers, be in the state’s different vegetation types?

Our simulation modelling suggests that, under dangerous fire-weather conditions that occur typically three to four times each summer, the geographic pattern in maximum fire intensity across Tasmania is most strongly shaped by vegetation type and fuel load, and to a lesser extent, slope and aspect. That fire intensity is controlled by these variables reflects the McArthur fire behaviour equations that underpin the Phoenix model (Gill et al. 1987). One reason why vegetation was of paramount importance in affecting fire intensity (Fig. 4b, Table 4) is because the McArthur model predicts rate of spread as a linear function of FFDI and fuel load (McArthur 1973; Noble et al. 1980). By extension, intensity is calculated as a linear function of FFDI (which we controlled) and fuel load squared (Gill et al. 1987). Thus, the fire intensity outputs in our modelling are more sensitive to fuel load (and hence vegetation type) (Bradstock 2010). This is a flaw of the McArthur model, as empirical studies have suggested the opposite is true, that fire severity is more sensitive to weather, especially as the fire weather becomes more severe (Price and Bradstock 2012).

The negligible effect of regional fire weather identified by the modelling might partially be a result of our constraint of using weather inputs derived only from 99th to 99.5th percentile FFDI days across Tasmania (8.7–38 FFDI). Such FFDI values are low compared with mainland southern Australia where the 99th percentile values can reach the high 50s in dry eucalypt woodlands (Price et al. 2015a). It should also be noted that these FFDI values are substantially less than the values of 100+ observed during some of Australia’s worst fire disasters (Engel et al. 2013). Still, our range of FFDI values represents a substantial variation in moderate and high fire-danger days, so it is surprising that our fire intensity GLMs are not more sensitive to FFDI, given the primary importance of fire weather identified by previous research (Penman et al. 2013; Price et al. 2016). However previous simulation studies did not analyse such a diverse suite of vegetation as exists across Tasmania, and this result likely reflects the substantial differences in fuel load between Tasmania’s vegetation types.

Of the five vegetation types used in this study that were considered completely untreatable with prescribed burning, four (highland vegetation, rainforest, wetlands and other environments) supported fires mostly or entirely below the 10 000 kW m\(^{-1}\) threshold at which firefighting must be curtailed, the other (wet eucalypt forest) was found to sustain enormously intense fires. Although the actual energy released from these fires has not been accurately determined, they can be very intense (>35 000 kW m\(^{-1}\)), often causing stand-replacing fires (Ashton 1976). This study suggests that the concentration of wet forests in the state’s south-east, including the surrounds of the capital, Hobart, could result in extremely high fire intensities, regardless of fuel treatment (Fig. 2). These forests are much less well understood than their Victorian counterparts, and there is inadequate data to build fuel accumulation curves for Tasmania’s wet eucalypt forests. As a result, we had to use the default Phoenix fuel accumulation curves that were fitted with data from Victorian forests, despite their fundamentally different stand structures (Turner et al. 2009).

Even though the maximal prescribed burning scenario is not operationally implementable, understanding its effects is important, as it represents an upper bound on the capability of prescribed burning in Tasmania. Though it could never be implemented on a state-wide scale, similar treatments could be carried out at much smaller scales, such as around the perimeter of human settlements. We found that this maximal scenario had two effects. The first was that this scenario reduced the probability of burning in every vegetation type (except highland vegetation). This is an important result, as it indicates that a maximal prescribed-burning regime has the ability significantly reduce the extent of fire, even in landscapes considered untreatable. The pattern of landscape fires has been shown to be strongly shaped by vegetation type in Tasmania, which creates self-reinforcing vegetation mosaics (Wood et al. 2011). Such mosaics are important for fire management because it is possible to manage fuel loads in flammable vegetation, such as sedgeland, thereby providing some protection from wildfire occurrence in either vegetation that supports uncontrollable conflagrations, such as wet eucalypt forests, or in vegetation that is highly fire sensitive, such as rainforests, and can be destroyed by wildfire (e.g. King et al. 2008). This effect explains why our study found that maximal prescribed burning significantly reduced the probability of fire presence in untreated vegetation (Fig. 4, Table 4). This has already been shown in sedgelands with prescribed burning protecting fire sensitive rainforest (King et al. 2008).

The second effect of the maximal prescribed-burning scenario was that large reductions in maximum intensity only occurred in the three vegetation types that are considered to be extensively treatable through prescribed burning (dry eucalypt
forest, sedgelands and native grassland; Fig. 4). This suggests, that even under a best-case scenario, prescribed burning does not have the ability to reduce fire intensities in vegetation that is not treated. This is especially problematic in wet eucalypt forests, where prescribed burning is not possible and fires have the potential to burn at extremely high intensities. This result underscores the need to investigate alternative fuel treatments in such vegetation where prescribed burning is not possible, especially wet eucalypt forests, for some type of fuel reduction treatment will be necessary to reduce fireline intensities.

Although the maximal scenario provides important insights into the upper bounds of the ability of prescribed burning to reduce fire activity, and may provide some insights into the ecological effects of intensive, landscape-wide burning practiced by Aboriginal peoples for millennia (e.g. Gammage 2011), it is not a scenario that could ever be implemented by fire managers. Therefore we analysed the effects of 12 more implementable, hypothetical prescribed-burning plans, designed by fire managers to treat large areas of landscape (see Table S1). We found that these more realistic, implementable plans, were much less effective in reducing fire extent and intensity when compared with the maximal scenario. To analyse these implementable plans, we applied the concept of ‘leverage’ (Loehle 2004) that is typically used to analyse the effectiveness of prescribed burning with respect to reducing the extent of wildfire (Price et al. 2015a). We found that these implementable prescribed-burning plans exhibited an ability to reduce area burnt (i.e. leverage) in only two of the three entirely treatable vegetation types (dry eucalypt forest and native grassland, but not sedgelands), and in two of the partially treatable vegetation types (native scrublands and non-eucalypt forest) (Fig. 5). Our estimate of 0.27 for dry eucalypt forest compares with estimates of 0.15–0.3 based on empirical (Price and Bradstock 2011; Price et al. 2015a) and simulation (Bradstock et al. 2012) studies in south-eastern mainland Australia and 0.25 in the south-west Australian dry eucalypt forests (Boer et al. 2009). There are no previous estimates for temperate Australian grasslands; however, area-burnt leverage in the extensive north Australian savannas has been estimated to be much higher at 0.9 than our estimate of 0.28 (Price et al. 2012). It is possible that this discrepancy relates to the strong contrasting spatial scale of these vegetation types, given that temperate grasslands are now heavily fragmented following agricultural development (Romain et al. 2016).

These 12 implementable prescribed-burning plans treated substantially smaller areas of the state than the maximal scenario, and the effectiveness of prescribed burning under these implementable scenarios was considerably less. For instance, in vegetation types exhibiting area-burnt leverage, our leverage values associated with the implementable treatments ranged from 0.2 to 0.28, compared with an estimated leverage of 0.5 to 1 associated with the maximal scenario. Similarly, our intensity leverage estimates suggest the implementable prescribed-burning plans have a limited aggregate effect on intensity. To put our estimates in perspective, it would take ~100 000 ha of treatment in grasslands and 175 000 ha in dry eucalypt forest to reduce the average worst-case fire intensity by 1500 kW m⁻¹. This suggests that the efficacy of these hypothetically implementable prescribed-burning plans in reducing the extent and intensity of wildfire is relatively limited compared with the maximal treatment.

Despite a substantial effect of the maximal prescribed-burning scenario on fire activity in sedgelands, this vegetation type did not exhibit area-burnt leverage when we analysed the 12 implementable prescribed-burning plans. This difference appears related to the effects of fuel accumulation and the spatio-temporal pattern of treatment in these implementable plans. The treatments designed for sedgelands in these plans are characterised by generally large treatment blocks (see Table S1), or by a tendency to be clumped, generally around the infrastructure in the state’s west (Richards et al. 2014). These two characteristics had separate but similar effects on fire presence. Although spatially clumped treatment blocks may have been effective at protecting human infrastructure, they represent a small treatment area, and hence a low probability of a wildfire encountering them. At the same time, the extremely large treatment blocks seem to be especially ineffective in reducing fire spread and exert an outsized influence on the area-burnt leverage calculations in sedgelands (see Fig. S1). This is likely related to the sedgelands’ high fuel re-accumulation rates (Marsden-Smedley and Catchpole 1995), as large treatment block sizes will result in large areas of even-aged fuels, and the fast re-accumulation rates will provide an unimpeded fuel load on which fires can spread. Indeed, area-burnt leverage has been shown to be negatively related to fuel accumulation rates (Price and Bradstock 2012; Price et al. 2015a). If we remove these large treatments from our analysis (2017 and 2018 in plan 5 and 2017 in plan 9; see Table S1), our leverage estimates increase to being slightly positive. This suggests that the effectiveness of prescribed burning in Tasmania’s sedgelands decreases as the homogeneity of fuel loads increases. Previous simulation research on south-west Tasmanian sedgelands has found the spatial patterning of prescribed burning to be critically important in explaining fire activity (King et al. 2006) and that leverage decreases as the range of areas treated enlarges (King et al. 2008; King et al. 2013; Price et al. 2015b). Simulation research on generic, synthetic landscapes has indicated that a mosaic of small, overlapping patches is most effective in reducing area burnt and rate of spread (Finney 2001; Duncan et al. 2015). Indeed, this is an approach that most closely matches Aboriginal patterns of fire use (e.g. Trauernicht et al. 2015). However, there is little empirical research on optimal treatment patterns, and further research is required to understand how fire spread and intensity is affected by the pattern of burning in Tasmania’s flammable sedgelands (e.g. French et al. 2016).

In this study we also applied the leverage concept to reductions in fire intensity. We found that an extensive amount of prescribed burning is necessary to reduce the highest fire intensities on a bad fire-weather day (Fig. 5). Even though previous studies have found specific treatments to be effective at reducing fire intensity and aiding suppression when a fire encounters a treatment (Billing 1981; Grant and Wouters 1993; McCarthy and Tolhurst 2001), our study indicates that the aggregate effect of this across the island of Tasmania will be quite low, as it would only take one fire encountering an untreated area to produce a potentially intense conflagration. Indeed, previous simulation studies have suggested that
prescribed burning is less effective at reducing intensity than it is at reducing area burnt (Bradstock et al. 2012). However, empirical studies have shown fuel load to be among the most important predictors of satellite-derived fire severity (which is correlated with intensity) in dry eucalypt forests (Bradstock et al. 2010) and of field-derived fire severity in Tasmania’s sedgelands (French et al. 2016). This would suggest that prescribed burning can exhibit intensity leverage, and that further empirical study is required. Additionally, as mentioned earlier, our intensity-leverage metric does not account for the spatial patterning of burning, which has been shown to significantly affect fire propagation in Tasmanian sedgelands (King et al. 2008), and more generally reduce rates of spread (and hence intensities) in wildfire (Finney 2001). Lastly, our pragmatic use of a uniform distribution of ignitions in the simulations may have resulted in an under-prediction of intensity leverage. In reality, managers can concentrate fuel treatments around areas with a high likelihood of ignition. This should result in prescribed burning being more effective at reducing aggregate intensities.

An ordination of both area-burnt and intensity leverage estimates indicates that trends in area-burnt and intensity reductions associated with prescribed burning are fundamentally different among the five vegetation types for which we calculated leverage (Fig. 5). The result suggests that the effectiveness of the implementable prescribed-burning plans differs among these vegetation types, with the greatest effect in native grassland, and the least in sedgelands. However, more research is needed on the potential for fine-scale mosaic treatments in altering fire behaviour, especially in this latter vegetation type.

Our study presents repeated simulations of a fire behaviour model across an entire island of some 68 000 km², as well additional 12 000 km² of outlying islands. This is much broader in geographic scale and ecological diversity than previous related published simulation studies (Ager et al. 2007; Beverly et al. 2009; Carmel et al. 2009; Penman et al. 2013). However, we acknowledge our approach was based on several assumptions.

First, we simulated ignitions with a uniform distribution across Tasmania, although in reality fires are more likely to start around population centres. This was a practical choice given the computational and data requirements for simulating from an empirical ignition distribution at a regional scale, but we recognise there are limitations to this approach. Importantly, one-third of the implementable plans were designed to exclusively target areas directly surrounding human settlements. By using a uniform ignition grid and saturating the landscape with fire, we nullify the advantages of such an approach, as in reality fires are more likely to start around human settlements, which should theoretically result in high leverage values associated with these treatments. Indeed, using an empirical (as opposed to a random) ignition distribution has been shown to result in significant differences in simulated fire size and productivity, though this difference is less pronounced under extreme weather (Bar Massada et al. 2011). Further modelling, particularly at the local scale, should simulate ignitions by sampling from an empirical ignition distribution based on historical ignition records.

Second, our use of a single synthetic fire-weather day for each climate region, as well our limitation of daylength to 6 h, was a pragmatic decision given computing constraints. This limitation is problematic, especially in a topographically diverse landscape such as Tasmania, where a highly varied topography and steep elevational gradients result in large climactic variation in small areas. The limited daylength is less of a problem, because our saturation of the landscape with fires rendered the size of individual fires as less important, but still should be acknowledged. Future modelling should sample fire weather from meteorological observations, including the most extreme fire-weather conditions (namely well above the 99.5th percentile FFDI) that were excluded from this study, as well as from projected weather under climate change (Grose et al. 2010).

Lastly, the McArthur model (the basis for forest fire behaviour in Phoenix) was built based on over 800 observations of wildfire in the ‘high eucalypt forests’ of New South Wales (Noble et al. 1980) and the Phoenix model was parameterised using fuel curves describing many Tasmanian vegetation types for which empirical fuel loads have not been adequately measured. We suspect inadequate parameterisation of the model may account for the prediction of >200 000 kW m⁻¹ fire intensities in wet eucalypt forests, which is much higher than what is considered to be the maximum fire intensity possible in eucalypt forests (Gill and Moore 1990). Indeed, the Tasmania Fire Service works under the assumption that the Phoenix model generally over-predicts actual fire intensities in wet forests (R. Richards, Tasmania Fire Service, pers. comm., 29 October 2016). The difficulty in applying the Phoenix model to forest types other than those for which it was developed has been previously recognised: the McArthur model (which underpins Phoenix) has been found to under-predict rates of spread in Western Australian Jarrah forests, especially in extreme weather conditions and dense understoys (McCaw et al. 2008). The need to develop fire behaviour models suitable for extreme weather conditions has been identified as a top research priority in Australian emergency management (Bushfire and Natural Hazards CRC 2016). We appreciate that these extra layers of complexity are required for more thorough analysis. We recognise that our findings are preliminary, and are based on a pragmatic study design balancing complexity against practicality, but they are critical in contextualising fire management options in Tasmania.

In conclusion, our simulation study has shown that realistic, implementable prescribed-burning plans to reduce fine fuel loads in fire prone Tasmanian grasslands, sedgelands and dry eucalypt forests have little potential to substantially reduce the extent and intensity of wildfires at a state-wide scale. This would suggest that area-based prescribed-burning targets have little value without some sort of strategic implementation or risk-reduction framework, as an immense amount of burning would be required to achieve state-wide reductions in fire extent and intensity. In contrast, an impractical ‘maximal’ scenario shows that unrealistically intensive prescribed burning could reduce the extent and intensity of fires in these vegetation types, as well as the extent of fire in other untreated vegetation types. Although this maximal scenario could never be implemented at a state-wide scale (the implementable plans represent a range of ~1–25% of the area treated by the maximal scenario), its effectiveness suggests that intensive, localised prescribed-burning plans could be used for protecting assets or targeting...
areas of high risk. However, any benefits from such prescribed burning must be carefully weighed against the side effects of intensive prescribed burning on human health (e.g. Broome et al. 2016). Additionally, we suggest that this research demonstrates the need to investigate new fuel-treatment techniques, such as spatio-temporal patterns of prescribed burning designed to create fine-scale fuel mosaics, or general alternatives to prescribed burning such as mechanical thinning. That the effectiveness of prescribed burning varies greatly with vegetation type further emphasises the need for careful planning across the complex vegetation mosaics that characterise much of Tasmania. Improved design of prescribed-fire regimes will require improved fire-behaviour models, based on empirical measurement of fuel loads, particularly in wet eucalypt forests, and assessments of the effectiveness of prescribed burning in influencing the behaviour of actual wildfires (e.g. Price and Bradstock 2010; French et al. 2016).

Conflicts of Interest

The authors declare that they have no conflicts of interest.

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References

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Int. J. Wildland Fire M


McArthur AG (1973) ‘Forest Fire Danger Meter.’ (CSIRO: Canberra, ACT, Australia)


