Journal of Environmental Management 343 (2023) 118171

Contents lists available at ScienceDirect

Journal of Environmental Management

journal homepage: www.elsevier.com/locate/jenvman

Research article

Fuel reduction burning reduces wildfire severity during extreme fire events in south-eastern Australia

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ARTICLE INFO

Keywords:
Eucalypt forest
Burn patchiness
Extreme fire events
Fire hazard
Fuel reduction burning
Prescribed burning
Wildfire severity

ABSTRACT

Extreme fire events have increased across south-eastern Australia owing to warmer and drier conditions driven by anthropogenic climate change. Fuel reduction burning is widely applied to reduce the occurrence and severity of wildfires; however, targeted assessment of the effectiveness of this practice is limited, especially under extreme climatic conditions. Our study utilises fire severity atlases for fuel reduction burns and wildfires to examine: (i) patterns in the extent of fuel treatment within planned burns (i.e., burn coverage) across different fire management zones, and; (ii) the effect of fuel reduction burning on the severity of wildfires under extreme climatic conditions. We assessed the effect of fuel reduction burning on wildfire severity across temporal and spatial scales (i.e., point and local landscape), while accounting for burn coverage and fire weather. Fuel reduction burn coverage was substantially lower (~20–30%) than desired targets in fuel management zones focused on asset protection, but within the desired range in zones that focus on ecological objectives. At the point scale, wildfire severity was moderated in treated areas for at least 2–3 years after fuel treatment in shrubland and 3–5 years in forests, relative to areas that did not receive fuel reduction treatments (i.e., unburnt patches). Fuel availability strongly limited fire occurrence and severity within the first 18 months of fuel reduction burning, irrespective of fire weather. Fire weather was the dominant driver of high severity canopy defoliating fire by ~3–5 years after fuel treatment. At the local landscape scale (i.e., 250 ha), the extent of high canopy scorch decreased marginally as the extent of recently (<5 years) treated fuels increased, though there was a high level of uncertainty around the effect of recent fuel treatment. Our findings demonstrate that during extreme fire events, very recent (i.e., <3 years) fuel reduction burning can aid wildfire suppression locally (i.e., near assets) but will have a highly variable effect on the extent and severity of wildfires at larger scales. The patchy coverage of fuel reduction burns in the wildland-urban interface indicates that considerable residual fuel hazard will often be present within the bounds of fuel reduction burns.

2. Introduction

Extreme wildfire events are increasing across forested regions worldwide (Abatzoglou and Williams, 2016; Collins et al., 2022; Hanes et al., 2018), leading to devastating social and environmental outcomes (Nolan et al., 2021; Ward et al., 2020). Warmer and drier conditions caused by anthropogenic climate change are increasing forest flammability, elevating the likelihood of large, high-intensity wildfires (Abatzoglou and Williams, 2016; Collins et al., 2022). Fuel management practices and land use change have contributed to increased fire activity in some regions (e.g., western USA, Mediterranean Europe), exacerbating the effects of climate change on fire regimes (Abatzoglou et al., 2021; Pausas and Fernández-Muñoz, 2012). Mitigating the impacts of extreme wildfires arising under climate change is a major challenge for fire management agencies (Geary et al., 2022; Nolan et al., 2021).

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https://doi.org/10.1016/j.jenvman.2023.118171
Received 8 March 2023; Received in revised form 10 May 2023; Accepted 12 May 2023
Available online 26 May 2023
0301-4797/

Abbreviations

APZ	Asset Protection Zone
BMZ	Bushfire Moderation Zone
LMZ	Landscape Management Zone
CFM	canopy foliar moisture
FFDI	Forest Fire Danger Index
FRB	fuel reduction burn
RFRB	recent fuel reduction burning
SMR	root zone soil moisture
TPI	topographic position index
TSFR	time since fuel reduction burning
UB	unburnt
LCS	low canopy scorch
MCS	moderate canopy scorch
HCS	high canopy scorch
CB	canopy burnt

Prescribed burning with the objective of fuel hazard reduction, referred to hereon as fuel reduction burning, is widely used across forests, woodlands, shrublands and grasslands for the mitigation of wildfire impacts on assets (e.g., life and property, timber, water supply; Fernandes and Botelho, 2003; Penman et al., 2011). It aims to reduce the biomass and connectivity of fine fuels (<6 mm diameter), thus moderating future wildfire behaviour (i.e., rate of spread, intensity, spotting), and increasing the likelihood of fire suppression (Fernandes and Botelho, 2003, Penman et al., 2020). In practice, the goal of a fuel reduction burning program may range in scale from targeted treatments aimed at moderating fire behaviour near assets, to landscape-scale treatment(s) aimed at limiting the growth of wildfires (McCaw and Burrows, 2020). The effectiveness of fuel reduction burning will vary across space and time, depending on the degree of fine fuel modification (i.e., biomass consumed, area treated), the re-accumulation of fuels over time, the likelihood a wildfire reaches the treated area, and the environmental conditions when a wildfire encounters the fuel treatment (Davim et al., 2021; Fernandes and Botelho, 2003; Penman et al., 2020).

Increased fuel reduction burning has been proposed as a strategy to help combat the elevated risk of extreme wildfire events associated with climate change (e.g., Morgan et al., 2020). This includes calls for holistic burning programs aimed at mitigating the impact of extreme fire events on both communities and ecosystems (Bowman et al., 2020a; Stephens et al., 2020), along with the integration of traditional knowledge and practices, including cultural burning (Fletcher et al., 2021). In temperate forest biomes, large wildfire occurrence, size, and severity are primarily driven by fire weather and drought, with fuel biomass having secondary effects on fire size and behaviour (Abatzoglou et al., 2018; Collins et al., 2022). The effectiveness of fuel reduction burning will diminish in temperate forests if the top-down effects of climate change (i.e., increased drought and severe fire weather) (Abram et al., 2021) override the influence of fuel reduction (Clarke et al., 2022). Recent fire seasons in which extreme fire events occurred, provide a unique opportunity to examine the efficacy of contemporary fuel reduction burning under extreme climatic conditions.

The 2019-20 Australian 'Black Summer' wildfires were unprecedented in terms of duration and extent, with ~7.5 million hectares of forest burning across south-eastern Australia between August 2019 and March 2020 (Bowman et al., 2020b; Davey and Sarre, 2020). High severity fire was widespread, with ~44% of native woody vegetation across the burned areas experiencing complete canopy defoliation (i.e., complete scorch or consumption; Bowman et al., 2021; Collins et al., 2021). Although it has been established that antecedent drought and fire weather were the primary drivers of this extreme fire season (Abram et al., 2021; Collins et al., 2022), debate around the contribution of ecosystem management to the size and impact of these wildfires has been ongoing (e.g., Bowman et al., 2021; Bradstock et al., 2020; Lindenmayer et al., 2022). To date, analysis of the effect of forest management on the Black Summer fires has primarily focused on timber harvesting (Bowman et al., 2021; Lindenmayer et al., 2020), with less focus on the effect of fuel reduction burning (although see Hislop et al., 2020; Nolan et al., 2021).

Fuel reduction burning is used extensively across south-eastern Australia for fuel hazard reduction (McCarthy et al., 2017; Penman et al., 2020). The effect of fuel reduction burning on fine fuel biomass in eucalypt forests, woodlands and shrublands is well established: burning decreases fine fuels below 'hazardous' levels (i.e., ~10 t/ha), with rapid re-accumulation occurring post-burn, such that steady-state biomass is attained within 5–15 years (Penman et al., 2020). The spatial coverage of treated fuels within the perimeter of fuel reduction burns (i.e., 'burn coverage') is poorly understood in the Australian context, despite being an important indicator of successful risk reduction (McCaw and Burrows, 2020). The limited data available indicates that burn coverage is variable, ranging from 5% to 90%, with a mean coverage of 40% for small experimental burns (<100 ha) (Penman et al., 2007), and 25% for larger management burns (>500 ha) (McCarthy et al., 2017).

Research testing the effect of fuel reduction burning on wildfire behaviour in southern Australia has primarily focused on relationships between remotely sensed metrics of fire severity and time since previous fire, irrespective of fire type (Penman et al., 2020). Fire severity measures changes in above- and below-ground organic matter resulting from fire (Keeley, 2009). In Australian forests, fire severity is quantified by the degree of foliage scorch and consumption (e.g., McCarthy et al., 2017), which is a function of fire behaviour (i.e., fire intensity, flame height) and the structural characteristics of the stand (Burrows, 1997). Past research has shown that fire severity is typically lower in young fuel ages (<5 years) compared to older fuels (e.g., >15 years since fire), though this fuel age effect tends to be overriden by severe fire weather (Penman et al., 2020). Studies specifically targeting the effect of fuel reduction burning on wildfire severity are limited (e.g., Hislop et al., 2020; Tolhurst and McCarthy, 2016) and do not account for the internal patchiness of burns. Consequently, these analyses typically underestimate the effect of treated fuels on wildfire severity. Furthermore, most studies have examined wildfire severity as a unitless metric (e.g., difference Normalised Burn Ratio; Hislop et al., 2020) or binary classes (e.g., high vs low severity; Bowman et al., 2021), limiting the inference of findings from both a social and environmental perspective.

The 2019-20 Black Summer wildfires provide a unique opportunity to test the effect of fuel reduction burning in south-eastern Australia under extreme climatic conditions. Here, we use a database of burns with mapped internal (i.e., within perimeter) burn coverage to assess the effect that fuel reduction burning had on the severity of 10 wildfires that occurred under a range of conditions during the 2018-19 and 2019-20 fire seasons. Wildfires from the 2018-19 season were included to incorporate a broader range of climate conditions into the analyses (see 'Study area' description). Analyses were undertaken at both the point and landscape scale, with point scale analyses being used to quantify the effect (i.e., magnitude and duration) of fuel treatment on wildfire severity (e.g., Price and Bradstock, 2012), and landscape-scale analyses used to quantify wildfire risk reduction (e.g., the extent of 'insuppressible' high severity wildfire; McCaw and Burrows, 2020). Ordinal and Dirichlet regression were used, allowing us to estimate the effect of the drivers of the likelihood (Ordinal) and proportion (Dirichlet) of five discrete fire severity classes. We tested the following questions:

(1) how effective are fuel reduction burns at treating fuels within the planned burn area (i.e., how complete is the burn coverage);
(2) to what degree does fuel reduction burning moderate wildfire severity during extreme fire seasons and for how long does the effect persist; and,

(3) to what extent is the effectiveness of a fuel reduction burning contingent on the weather conditions at the time of the wildfire?

3. Methods

3.1. Study area

The study focused on temperate forests and shrublands in the Gippsland and Hume regions of eastern Victoria, Australia (Fig. 1). Wildfires burnt ~2 million hectares of native vegetation across the study area between January 2019 to April 2020, with >1.5 million hectares being affected during the 2019–20 'Black Summer' (Geary et al., 2022). We targeted 10 wildfires (referred to hereon as 'study wildfires') that occurred within areas that had been subjected to fuel reduction burning for wildfire risk reduction (Fig. 1). Six wildfires occurred during the 2018-19 fire season and four fires occurred during the 2019-20 fire season. The size of the wildfires ranged from ~1700 ha–1.2 million ha, with average size being larger in 2019–20 (mean = 459,300 ha, range = ~44,300 ha–1.2 million ha) than 2018–19 (mean = 22,600 ha, range = ~1700 ha–81,000 ha).

The study area falls in a temperate climate zone with mean annual precipitation ranging from 500 mm to 2200 mm and mean daily maximum temperature ranging between 16 °C and 30 °C in summer (www.bom.gov.au, accessed 8th July 2020). Elevation ranges from sea level to ~2000 m, with lower temperatures and higher rainfall being experienced in coastal and mountainous areas. The study area experienced widespread drought in the 12 months prior to the 2018-19 fire season, with severe rainfall deficiencies (rainfall <10th - 30th percentile) being recorded across eastern Victoria (www.bom.gov.au, accessed 8th July 2020). Severe drought conditions continued into the 2019-20 fire season, with much of the study area experiencing the lowest rainfall over a 24-month period on record (www.bom.gov.au, accessed 8th July 2020).

Vegetation across the study area is a mosaic of native eucalypt forest, rainforest, shrubland and grassland, intermixed with agricultural areas and urban settlements. Native woody vegetation communities in the region can be divided into five broad groups: open forests, tall-open forests, ash forests, rainforests, and shrubland. Open and tall-open forests are the dominant native vegetation types across the study area. Both forest types have an open canopy layer (30%–70% cover), with open forests being characterised by trees that are 10–30 m tall, and tall-open forests by trees that are >30 m tall. Open forests occur in soils of low to moderate fertility whereas tall-open forests occupy areas of greater soil fertility and/or relatively high precipitation (e.g., annual rainfall ≥1000 mm) (Cheal, 2010; Gill and Catling, 2002). Both forest types are dominated by eucalypts that can resprout epicormically following high severity fire. Ash forests are a special class of tall open forest (canopies up to 90 m) that occur in cool montane sites of high productivity (annual rainfall >1000 mm) and are dominated by obligate-seeder eucalypts. Rainforests are dominated by non-eucalypt tree species (10–25 m tall, >70% canopy cover), such as *Elaeocarpus holopetalus*, *Atherosperma moschatum* and *Acacia melanoxylon*, and occur in areas of high annual rainfall on sites that are protected from fire (e.g., deep gullies) (Cheal, 2010; Keith, 2004). Shrublands occur in areas of shallow soil with low nutrients and consist of a canopy layer dominated by shrubs (<10 m tall), with few trees or grasses present (Cheal, 2010; Keith, 2004).

The flammability of temperate forests in south-eastern Australia is strongly constrained by fuel moisture (Nolan et al., 2016), with drought being an important precursor for large wildfires (Bradstock, 2010). Drought severity is a critical determinant of the likelihood that a vegetation community will burn. For example, large areas of mesic forest communities (i.e., ash forest, rainforest) typically only burn during very severe drought (Cheal, 2010; Collins et al., 2019). Wildfires are of mixed severity with high severity fire (i.e., complete canopy scorch and/or canopy consumption) typically accounting for 30–50% of total wildfire extent (Collins et al., 2021). Wildfire severity patterns are strongly influenced by fire weather, with high severity fire dominating under conditions of severe to extreme fire weather (Bradstock et al., 2010; Price and Bradstock, 2012).

Fuel reduction burning is widely used across the study region to mitigate wildfire impacts and aid wildfire suppression (Penman et al., 2011). The protection of life, property and key assets (e.g., public infrastructure, water supply) are the primary focus of fuel management in the region, though consideration is also given to environmental values (e.g., biodiversity) (Penman et al., 2011). On public land in Victoria, fuel

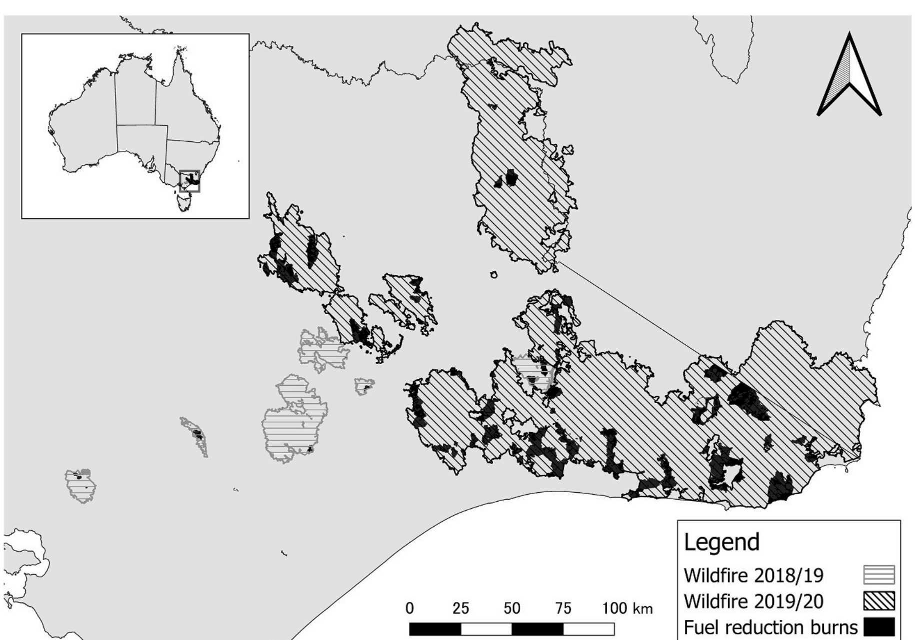

Fig. 1. The location of the study area in south-eastern Australia. The hatched polygons show the 10 study fires occurring in the 2019 and 2020 fire seasons. The black polygons show the location of the fuel reduction burns examined in the point scale assessment of wildfire severity.

reduction burning programs focus on strategic zones within the landscape. These include the: (i) Asset Protection Zone (APZ), which is located at the wildland-urban interface and aims to provide localised protection to people and assets by reducing radiant heat and ember exposure; (ii) Bushfire Moderation Zone (BMZ), which is located near the wildland-urban interface and aims to moderate wildfire spread, intensity and ember production; and (iii) Landscape Management Zone (LMZ), which occurs outside of APZs, BMZs, and burn exclusion zones and aims to reduce overall wildfire hazard and support other environmental objectives (Gippsland Strategic Bushfire Management Planning Working Group, 2020). The targets for fuel reduction burning vary across the zones, based on the desired management outcome(s). In the study area, guidelines prescribe burns with high burn coverage (80–100% burn coverage) at 5–8 year intervals for the APZ, moderate burn coverage (50–70% burn coverage) at 8-15 year intervals for the BMZ, and variable burn coverage (typically <50%) and variable intervals for the LMZ (Gippsland Strategic Bushfire Management Planning Working Group, 2020). Management burns in the study region primarily target open-forests, tall-open forests, and shrublands, as these communities are more fire prone and have greater tolerance to short-interval fires than mesic forest communities (i.e., ash forest, rainforest).

3.2. Wildfire severity mapping

Fire severity maps (30 m resolution) were created for the 10 study fires using Landsat imagery and a random forest classifier that has been extensively trained and validated for forest and shrubland communities across the study area (Collins et al., 2020). The random forest classifier was trained using data from 43 wildfires occurring between 2006 and 2019 across south-eastern Australia, including numerous 'mega-fires' (i.e., fires >100,000 ha) (Collins et al., 2020, 2021). Five fire severity classes that quantify the degree of scorch and consumption of foliage were mapped: (i) unburnt vegetation (UB); (ii) understorey burnt and less than 20% scorch in the upper canopy layer (low canopy scorch; LCS); (iii) between 20% and 80% scorch in the upper canopy layer (moderate canopy scorch; MCS); (iv) greater than 80% scorch in the upper canopy layer (high canopy scorch; HCS); and (v) canopy foliage mostly consumed (canopy burnt; CB). Independent cross validation has shown that the method has very high classification accuracy (global accuracy ~88%) for wildfires across the study area (Collins et al., 2020, 2021). The fire severity mapping procedure was implemented in Google Earth Engine (Gorelick et al., 2017).

Areas of non-woody vegetation (<10% woody cover) were masked from the severity maps using the woody vegetation cover dataset created by Hansen et al. (2013). Areas within 30 m of sealed roads and 90 m of powerlines were also masked out as they create areas of no fuel or low statured vegetation that can influence fire severity (LC pers. obs.). Spatial layers of roads and powerlines were obtained from Victorian Department of Energy, Environment and Climate Action (DEECA) and Geoscience Australia (www.ga.gov.au), respectively.

3.3. Fuel reduction burn mapping

A database of fuel reduction burn severity mapping for eastern Victoria was used for the assessment of both burn coverage and the effect of fuel reduction burning on wildfire severity. The database consisted of two classes of burn: fuel reduction burns and ecological burns. In our study, we did not distinguish between the two burn types, as ecological burns will ultimately reduce fuel loads (i.e., we considered both as a form of fuel reduction burning). Burn severity maps were derived using the manual interpretation and digitisation of high resolution (~15 cm) post-fire aerial imagery by experienced staff within DEECA (for details see McCarthy et al., 2017). The fire severity classes described above were used for the manual interpretation. Validation of these severity maps has shown very high accuracy (94%) when compared to field-derived ground-truthing (McCarthy et al., 2017).

A total of 192 fuel reduction burns identified as being of high accuracy were contained in the database. These fires occurred between the 2011/12 and 2019/20 fire seasons. The study wildfires were potentially influenced by 86 mapped burns (Fig. 1); these burns were used to assess burn effectiveness (i.e., magnitude and duration) at the point scale. The interval between the burns and the study wildfires ranged from ~6 months to 8 years, covering the period over which fuel reduction burns are expected to be effective in eucalypt forests (~5 years; Penman et al., 2020). Fuel reduction burn severity classes were reclassified as either 'burnt' or 'unburnt' for analysis, owing to the typically low severity of these management burns (i.e., burnt patches exceeding LCS were rare; LC pers obs). Shapefiles of the burns were converted to raster layers with a 30 m resolution.

The database of fuel reduction burns with severity mapping did not include all recent (≤5 years) burns that had interacted with the study wildfires. Additional mapping of burns was required to facilitate assessment of the effect of the extent of recent fuel reduction burning on wildfire severity at the local landscape scale. Landsat derived severity maps were generated for unmapped prescribed burns (fuel reduction or ecological) occurring since 2013 that had potentially interacted with the study wildfires (i.e., overlapped or fell within a 2 km buffer; n = 161). Maps were derived in Google Earth Engine using the method described above. The Landsat-derived fuel reduction burn mapping has been found to distinguish between burnt and unburnt areas with very high accuracy (~90%; Collins et al., 2020). The severity maps were converted to binary classes (i.e., burnt vs unburnt) and added to the burn database to derive landscape metrics of recent fuel reduction burning.

3.4. Environmental datasets

The environmental datasets and sampling approaches used in our study varied for the assessment of fuel reduction burn coverage and wildfire severity at the point and landscape scale (Table S1, Appendix 1). We provide an overview of the environmental datasets, below. Details of the use of these datasets in each analysis are outlined in Section 3.5. Preparation of the environmental datasets was undertaken using the 'raster' (Hijmans, 2020) package in R (R Core Team, 2020).

3.4.1. Wildfire history

Historic data of wildfire extent was acquired from DEECA (https://datashare.maps.vic.gov.au/). Shapefiles of the perimeters of wildfires occurring on public land in Victoria were available as far back as 1903, though the accuracy and completeness of records of this nature are questionable prior to the 1980s (LC pers. obs.). We used fire perimeter data to calculate time since the previous wildfire within the extent of each of the study wildfires. A raster layer of time since wildfire was generated at 30 m resolution across the extent of each of the study wildfires.

3.4.2. Daily fire weather and fuel moisture

Fire behaviour in forests and shrublands is strongly influenced by fire weather and dead and live fine fuel moisture content. We used several gridded datasets to account for the effect of these parameters on burn coverage and wildfire severity.

Surface fire weather was quantified using the McArthur Forest Fire Danger Index (FFDI) (Luke and McArthur, 1978). The FFDI was used operationally across Australian forests until September 2022 as an indicator of the potential for fire growth and the likelihood of suppression success (Luke and McArthur, 1978). The FFDI combines weather variables that influence fine dead fuel moisture (i.e., temperature and relative humidity) and fire spread (i.e., wind), as well as short-term (20-day) moisture deficiency in the deep duff and upper soil layers (i.e., 'drought factor') (Noble et al., 1980):

$$FFDI = 1.25 \times D \times \exp\left[\frac{T - R}{30.0} + 0.0234 \times W\right]$$

Where D is the drought factor, T is temperature, R is relative humidity and W is wind speed. Index values typically range between 0 and 100, but may exceed 100 under catastrophic conditions. Operationally, FFDI is communicated using six fire weather classes: low to moderate (0–11); high (12–24); very high (25–49); severe (50–74); extreme (75–99), and; catastrophic (≥100) (Clarke et al., 2022; Tolhurst and McCarthy, 2016). A gridded dataset (5 km resolution) of maximum daily FFDI (2010–2020) was obtained from the Australian Bureau of Meteorology and used to represent surface fire weather conditions in this study.

Soil moisture data was used to account for variation in ecosystem water availability. We used root-zone soil moisture (SMR) as it provides a relevant metric of the effect that drought has on water availability for trees and shrubs. A gridded dataset (5 km resolution) of daily root zone soil moisture (<1 m deep) was acquired from the Australian Water Outlook (https://awo.bom.gov.au).

Live foliar moisture in the canopy layer was estimated from the Moderate Resolution Imaging Spectroradiometer (MODIS) satellite imagery (500 m resolution), using an empirical model derived for forests, woodlands, and shrublands of south-eastern Australia (Nolan et al., 2016). The foliar moisture model predicts foliar moisture using the Visible Atmospherically Resistant Index, which is derived from the green, red and blue regions of the electromagnetic spectrum (Nolan et al., 2016). Estimates of canopy foliar moisture (CFM) were derived immediately (i.e., typically within 8-days) before the ignition date of the fire of interest. We did not derive estimates during the fire as smoke affects reflectance in the visible regions of the electromagnetic spectrum.

Hotspot detections were used to allocate the date of burn across locations within wildfire perimeters for the point- and landscape-scale analyses of wildfire severity. This allowed for the assignment of daily FFDI and SMR to locations across the study area. Hotspot data was derived by combining thermal detections recorded from the MODIS and Visible Infrared Imaging Radiometer Suite (VIIRS) satellites, resulting in a product with a spatial resolution of ~500 m with sub-daily return time (https://firms.modaps.eosdis.nasa.gov/download/). Investigation of the hotspot data indicated that the detection of thermal anomalies can occur a considerable time after the passage of the active fire front. For example, extensive hotspot detection was recorded between midnight and 8 a.m. following major fire runs that occurred during the afternoon of the previous day (LC per. obs.). We considered all hotspot observations occurring before 9 a.m. as having burnt on the previous day.

3.4.3. Topography and vegetation

A digital elevation model (30 m resolution) derived from the Shuttle Topographic Radar Mission (Farr et al., 2007) was used to generate topographic variables (Table S1, Appendix 1), which are known to affect fire severity in eucalypt forests and woodlands (Bradstock et al., 2010; Price and Bradstock, 2012). Raster layers of elevation (m), slope (°) and aspect (°) were extracted from Google Earth Engine. Aspect was recalculated as aspect relative to north (ASPN), by subtracting 360° from all aspects greater than 180°, and returning the absolute difference. ASPN ranges from 0° to 180°, with values close to zero representing the drier northern aspects and values approaching 180° representing moist (south) pole-facing aspects (Nyman et al., 2015). Topographic position index (TPI) was calculated as the difference between a focal pixel and the mean of all pixels within a surrounding 500 m radius. We used a 500 m radius focal area as this has been found to be suitable for characterising topographic position within the study area (Collins et al., 2019).

A shapefile of ecological vegetation divisions and fire groups (units of vegetation classification commonly applied in the study region) was acquired from DEECA and used to assign vegetation into the five groups described previously: open forests, tall-open forests, ash forests, rainforests and shrubland (Table S1, Appendix 1). A raster layer was generated at 30 m resolution across the extent of each of the study wildfires.

3.5. Data sampling and analysis

3.5.1. Fuel reduction burn properties

The spatial properties of fuel reduction burns were examined using the fuel reduction burn database (see Section 3.3). We targeted burns that were classed as having high accuracy and were successfully ignited over some part of the planned burn extent ($n = 192$). For each burn, the area of burnt and unburnt vegetation within the planned extent of the burn (i.e., burn extent) was calculated and used to derive burn coverage (i.e., burnt area/burn extent). Burns were assigned to one of three fuel management zones (described above) based on mapping available from DEECA (https://discover.data.vic.gov.au/dataset/fire-management-zones2): (i) Asset Protection Zone (APZ), (ii) Bushfire Moderation Zone (BMZ), or (iii) Landscape Management Zone (LMZ). Fuel management zone data for the study region was current as of July 2020 and based on the state-wide 2017-19 Strategic Bushfire Management Planning process. If a burn occurred in more than one zone, it was assigned to the zone that covered the largest proportion of the burn extent.

Analysis of the drivers of burn coverage was undertaken at the burn scale (i.e., burns were the unit of replication). The average FFDI and SMR across the burn extent was extracted for the day that the planned burn was conducted, to account for temporal variation in fire weather conditions and soil moisture. The standard deviation of elevation across the burn extent was extracted to account for topographic heterogeneity. Riparian forests, ash forests and rainforest are generally not flammable during the prescribed burn window for the study area (i.e., Autumn, Spring), owing to their moist microclimates (Cheal, 2010). The coverage (%) of riparian forests, ash forest and rainforest, referred to collectively as ‘wet forests’ hereon (Table S1, Appendix 1), was calculated for the burn extent to account for the extent of vegetation with low ignitability. The area of the burn extent was also calculated. Data extraction from spatial datasets was undertaken using the ‘terra’ package (Hijmans, 2022) in R (R Core Team, 2020).

Bayesian regression implemented using the ‘brms’ package (Bürkner, 2017) in R was used to examine the factors influencing burn coverage. Models were fit using a beta distribution to account for the bounded nature of the response variable (i.e., between 0 and 1). In instances where burn coverage equalled 1 ($n = 4$), a small value (i.e., 10^{-6}) was subtracted from the burn coverage to meet the assumptions of the beta distribution (i.e., $0 < y < 1$). The FFDI, SMR, topographic heterogeneity, wet forest extent and burn extent were included as predictors in the model. Topographic heterogeneity, wet forest extent, and burn extent were log transformed. Predictors were normalised by subtracting the mean and dividing by the standard deviation. The interaction between FFDI and burn extent was also included in the model, as the sensitivity of burn coverage to burn extent should decrease with increasing severity of fire weather. The precision parameter for the model was linked to burn extent, to account for the greater variation in burn coverage for smaller burns. A summary of the model parameters is presented in Table S2, Appendix 1.

We used default priors and fit four chains with a burn-in of 1000 and a total of 5000 iterations. We used R-hat values and traceplot figures to check parameter convergence (Gelman et al., 2020). We used posterior predictive checks to verify that our models were able to reproduce the distribution of observed burn coverage.

3.5.2. Point-scale effect of fuel reduction burning

The point-scale assessment of fuel reduction burn efficacy focused on areas within the burn extent that were either burnt (treatment, $FRB = 1$) or unburnt (control, $FRB = 0$). A 40 m internal buffer was applied to the burnt and unburnt polygons to ensure that the sampled wildfire severity pixels (30 m resolution) fell entirely within the mapped treatment. A topographic landforms layer (http://developers.google.com/earth-engine/datasets/catalog/CSP_ERGo_1_0_Global_SRTM_landforms), which identified ridges, upper slopes, lower slopes and valleys, was used to stratify sample points across topography within the fuel reduction burns. Sample points were randomly generated within the polygons, with a minimum spacing of 200 m. Sample points greater than 1000 m from a wildfire hotspot detection were not considered in the analysis, due to reduced confidence about the date of burn. We also excluded sample points that were: (i) within 100 m of non-woody vegetation; (ii) greater than 100 m from a pixel that had been burnt by the study wildfires (to avoid sampling unburnt patches that were not exposed to wildfire); and (iii) had been recently harvested (<20 years) prior to the wildfire(s) of interest, as the probability of high severity fire is elevated during this period (Price and Bradstock, 2012). Environmental data was extracted for each point using the 'raster' (Hijmans, 2020) package in R. Fire weather was assigned to each point using the closest hotspot. Our analysis focused on points that occurred within the shrubland, open-forest, and tall-open forest communities (n = 7012), which are the focus of fuel reduction burning efforts in the study area. The sample points occurred across 85 prescribed burns and all 10 wildfires.

We used ordinal regression (Bürkner and Vuorre, 2019), fitted using the 'brms' package, to model the effect of a binary fuel reduction burn predictor (FRB; unburnt = 0, burnt = 1), time since fuel reduction burn (TSFR) and environmental predictors (i.e., FFDI, CFM, SMR, aspect, slope, TPI) on wildfire severity (an ordinal response variable with five ordered severity classes). We included an FRB identifier as a random effect to account for the fact that observations within the same FRB were not independent. We included the interaction between FFDI and FRB to quantify whether fire weather was modulating the protective effect of FRB on wildfire severity. The interaction between CFM and FFDI was also included because initiation of canopy fire is often contingent on severe fire weather. We used a multiple hypothesis approach (Hilborn and Mangel, 1997) to test five alternative functional forms for the interaction term between FRB and TSFR on wildfire severity (constant model, exponential decrease, delay decrease, saturating model, and optimal model). The five alternative ordinal models and model priors are described in Appendix 2. Models were fit with four chains (1000 iteration burn-in, 5000 iterations total) and were assessed using methods outlined in Appendix 2. Models that used an exponential decrease (wildfire severity is immediately reduced after FRB, but the protective effect of FRB decreases exponentially with TSFR) are presented as they produced the best fit. We focus on the effect of fuel reduction burning, FFDI, and moisture availability (CFM, SMR) when presenting and discussing the results of the analysis. Results related to the effect of topography are presented in Appendix 1 (Table S3 and Fig. S1).

We identified the minimum time frame over which fuel reduction burning was effective by calculating the youngest age when the 95% credible intervals for points treated with fuel reduction burning intercepted the 95% credible intervals for the untreated points for the UB, HCS and CB severity classes. We focused on these classes as they represent both a key objective of FRB, which is to prevent wildfire from reaching a severity that has a low likelihood of suppression (i.e., HCS, CB), and a possible outcome of FRB, the extinguishment of wildfire (i.e., UB). We identified the maximum time-frame over which fuel reduction burning was effective by calculating the youngest age when the predicted mean for treated areas intercepted the 95% credible intervals for untreated areas.

3.5.3. *Local-landscape scale assessment of fuel reduction burning*

A 250 ha (893 m radius) focal area was used for the local-landscape assessment of fuel reduction burn efficacy. This neighbourhood size was used because the dimensions were similar to the width of fire management zones that are focused on moderating fire behaviour at the wildland-urban interface (i.e., 500 m–2000 m). Landscapes were created by placing an 893 m buffer around the hotspots. We calculated the percentage of each landscape that had experienced fuel reduction burning up to five years prior to the target wildfire (i.e., recent fuel reduction burning; RFRB). Covariates related to fire weather and moisture included the FFDI, SMR, and average CFM across a 250 ha landscape. The FFDI and SMR were extracted for the pixel that intersected the hotspot, as the resolution of these grids (i.e., 5 km) was greater than the diameter of the 250 ha landscape. We calculated the percentage of wet forest and the standard deviation of elevation across each landscape to account for the extent of forest with low flammability and topographic heterogeneity, respectively. Raster processing and data extraction were undertaken using the 'raster' package in R.

For our analysis, we considered all landscapes associated with hotspots that fell inside a wildfire perimeter. This allowed for areas of the 250 ha landscapes to fall outside the wildfire extent. We incorporated this into our sampling design to account for the 'shadow effect' of fuel reduction burning, as recent fuel reduction burning may stop wildfire spread, through fuel limitation or by aiding suppression, protecting vegetation on the leeward side of the treatment. Landscapes were excluded if they contained any non-woody vegetation, sealed roads or powerlines, or had experienced wildfire in the previous five years or harvesting within the previous 20 years. Additionally, landscapes with >60% wet forest were excluded because these areas will have low levels of prescribed burning. Suitable landscapes were randomly subsetted so that there was a minimum distance of 2000 m between their mid-points, to ensure a >250 m spacing between landscape boundaries. Landscapes with some RFRB were sampled initially (n = 101). We then identified all landscapes with no RFRB that were greater than 2000 m and less than 5000 m from those with RFRB (n = 116), to derive a set of control points. The two data sets were then combined and used for the analysis. The sample points occurred across eight of the 10 wildfires.

Bayesian regression was used to examine the factors influencing the proportion of each severity class across a landscape. Models were fit using a Dirichlet distribution, which is suitable for proportional data with more than two levels. A very small area (i.e., 1 ha) was added to each severity class and the total landscape area was adjusted (i.e., 255 ha), prior to the calculation of the severity class proportions, to meet the assumption that proportions were greater than zero. RFRB, FFDI, SMR, CFM, topographic heterogeneity, and the percentage of wet forest were included as predictors in the model. A square root transformation was applied to RFRB and wet forest extent to reduce the skew in the distribution of these variables. The interaction between FFDI and RFRB was also included in the model, as the efficacy of recent burning was expected to decrease with increasing severity of fire weather. Predictor variables were normalised as described above. Models were fit with 4 chains and assessed using methods described in 3.5.1.

We focus on the effect of fuel reduction burning, fire weather, and moisture availability when presenting and discussing the results of the analysis. Results related to the effect of topographic heterogeneity and the percentage of wet forest are presented in Appendix 1 (Table S4 and Fig. S2).

4. Results

4.1. *Fuel reduction burn properties*

The average (±S.E.) size of the fuel reduction burns examined was 1990 ha ± 203 ha, with a range of 2–15527 ha (Fig. 2a). The average (±S.E.) burn coverage within the burn perimeter was 39.6% ± 2.1%, with a range from 0.1% to 100% (Fig. 2b). Burns conducted within the APZ were smaller (442 ha ± 70 ha) with greater coverage (55.4% ± 5.1%) than burns in the BMZ (1426 ha ± 124 ha, 39.3% ± 3.0%) or LMZ (3508 ha ± 489 ha, 32.2% ± 3.5%; Fig. 2). Burn coverage varied in response to SMR, the extent of wet forest types and the interaction between burn extent and FFDI (Table S2, Appendix 1). FFDI and burn extent had the largest effect on burn coverage (Fig. 3). Under 'low' FFDI (e.g., FFDI = 1), burn coverage decreased sharply with increasing burn extent, levelling off when burn extent exceeded ~2000 ha, with an ~80% decrease in burn cover across the range of burn extent (Fig. 3a). The effect of burn extent on burn coverage decreased with increasing FFDI, such that burn coverage was somewhat insensitive to burn extent

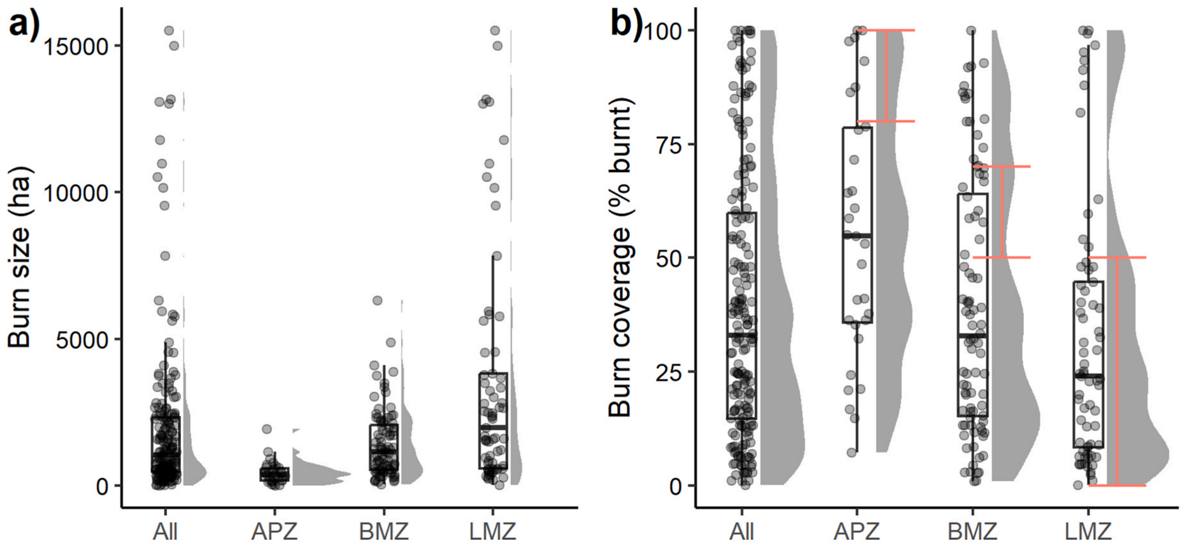

Fig. 2. Raincloud plots showing the distribution of a) burn size (ha) and b) burn coverage (% burnt) for the burns contained in the burn database. Plots are presented for all burns combined ('All'), Asset Protection Zones (APZ), Bushfire Moderation Zones (BMZ) and Landscape Management Zones (LMZ). The raincloud plots show the raw data (points), boxplots and a histogram. Boxplots depict the median (bold line), interquartile range (box) and 1.5x the interquartile range (whiskers). The red bars in b) indicate the target coverage for each fuel management zone.

under 'high' FFDI (e.g., FFDI = 12; Fig. 3a). Consequently, FFDI was most influential on the coverage of large burns (i.e., >2000 ha; Fig. 3a). There was a modest decrease in burn coverage (≈0.15) with increasing SMR and extent of wet forest (Fig. 3b and c).

4.2. Effect of fuel reduction burning at the point scale

Most sample points occurred in open forest (42.1%) and tall-open forest (43.9%). The LCS severity class was the most common (31.0%), followed by HCS (24.1%), MCS (21.6%), UB (14.3%) and CB (9.1%) (Table S5, Appendix 1). Wildfires were typically more severe in shrubland than open forest and tall-open forest (Table S5, Appendix 1). Areas that were treated (i.e., burnt) within the fuel reduction burn extent experienced lower wildfire severity on average than those areas that were not treated (i.e., unburnt) (Table S5, Appendix 1).

Wildfire severity patterns were affected by FFDI, FRB and TSFR in each vegetation community assessed (Appendix 1, Table S4), but not by variables related to water availability (i.e., SMR, CFM; Fig. 4). Wildfire severity increased with increasing FFDI, with similar trends being observed across each vegetation community (Fig. 4). Across the range of FFDI (i.e., 0 to 80), there was a decrease in the likelihood of UB ($\Delta P \approx$ 0.10) and LCS ($\Delta P \approx 0.25$) and an increase in the likelihood of HCS ($\Delta P \approx 0.20$) and CB ($\Delta P \approx 0.10$–0.20), with MCS remaining relatively constant ($P \approx 0.20$–0.25; Fig. 4). Under low FFDI (FFDI <12), there was predominantly low or moderate levels of canopy scorch ($P > 0.6$), whereas under severe to extreme FFDI (FFDI ≥50) there was an elevated likelihood ($P \approx 0.35$–0.7) of canopy defoliation (i.e., HCS or CB; Fig. 4). The FFDI at which there was a shift towards a greater likelihood of canopy disturbance (i.e., HCS, CB) than low canopy impact (i.e., UB, LCS) increased from shrubland (FFDI ≈ 20), open forest (FFDI ≈ 35), and tall-open forest (FFDI ≈ 50; Fig. 4). The UB and CB classes were rare under severe and low FFDI, respectively (Fig. 4).

Fire severity increased with increasing time since fuel reduction burning (TSFR) across all vegetation communities and levels of FFDI (Fig. 5). Across the range of TSFR (i.e., 0–8 years), there was a decrease in the likelihood of UB and an increase in the likelihood of MCS, HCS and CB (Fig. 5). The response of the LCS class varied with FFDI, either remaining relatively constant (FFDI = 12) or decreasing (FFDI ≥50) with increasing TSFR (Fig. 5). Fuel reduction burning effectively increased the likelihood of fires extinguishing (i.e., UB) and decreased the occurrence of fires that defoliate the canopy (i.e., HCS, CB) for up to 3–5 years in shrubland, 3–8 years in open forest and 5 to >8 years in tall-open forest (Fig. 6). Areas that were recently treated (i.e., TSFR <2 years) had a high probability of either not burning (i.e., UB) or experiencing low severity fire (i.e., LCS), across all communities and fire weather (Fig. 5). When TSFR exceeded 3 years, fire weather became the predominant driver of wildfire severity, and ultimately the likelihood of canopy defoliating fire (Fig. 5; Fig. 6). The slope of the relationship between TSFR and the likelihood of canopy defoliating fire severity classes (or alternatively, unburnt areas) became steeper under increasing FFDI and from taller to shorter statured vegetation (i.e., tall-open forest < open forest < shrubland; Fig. 5).

4.3. Effect of fuel reduction burning at the local-landscape scale

The proportion of each fire severity class within a 250 ha landscape was primarily influenced by FFDI. When FFDI increased, the proportion of UB and LCS decreased, while the proportion affected by HCS and CB increased (Fig. 7a). There was a transition towards a greater extent of canopy disturbance (i.e., HCS, CB) vs low canopy impact (i.e., UB, LCS) when FFDI exceeded ~30 (Fig. 7a).

RFRB primarily affected the proportion of LSC and HCS in the local landscape (Fig. 8). An increase in RFRB decreased the extent of HCS, and increased the extent of LCS, though the magnitude of change in the proportion of each severity class was relatively small considering the large increase in burn extent (e.g., HCS decreases by 0.10 in response to a 50% increase in RFRB; Fig. 8). There was considerable uncertainty around the effect of RFRB on both LCS and HCS, with 95% credible intervals displaying considerable overlap with the predicted mean across the range of RFRB (Fig. 8). RFRB had little effect on the other fire severity classes (Fig. 8).

A decrease in either SMR or CFM resulted in an increased contribution of high severity fire (i.e., HCS, CB) and decreased contribution of LCS, though the magnitude of change was often small ($\Delta Prop$ <0.1) relative to the uncertainty surrounding the parameter estimates (Fig. 7b and c).

5. Discussion

Despite the widespread application of fuel reduction burning across temperate woody ecosystems in south-eastern Australia, targeted assessment of the efficacy of this practice is limited. Our study is the first

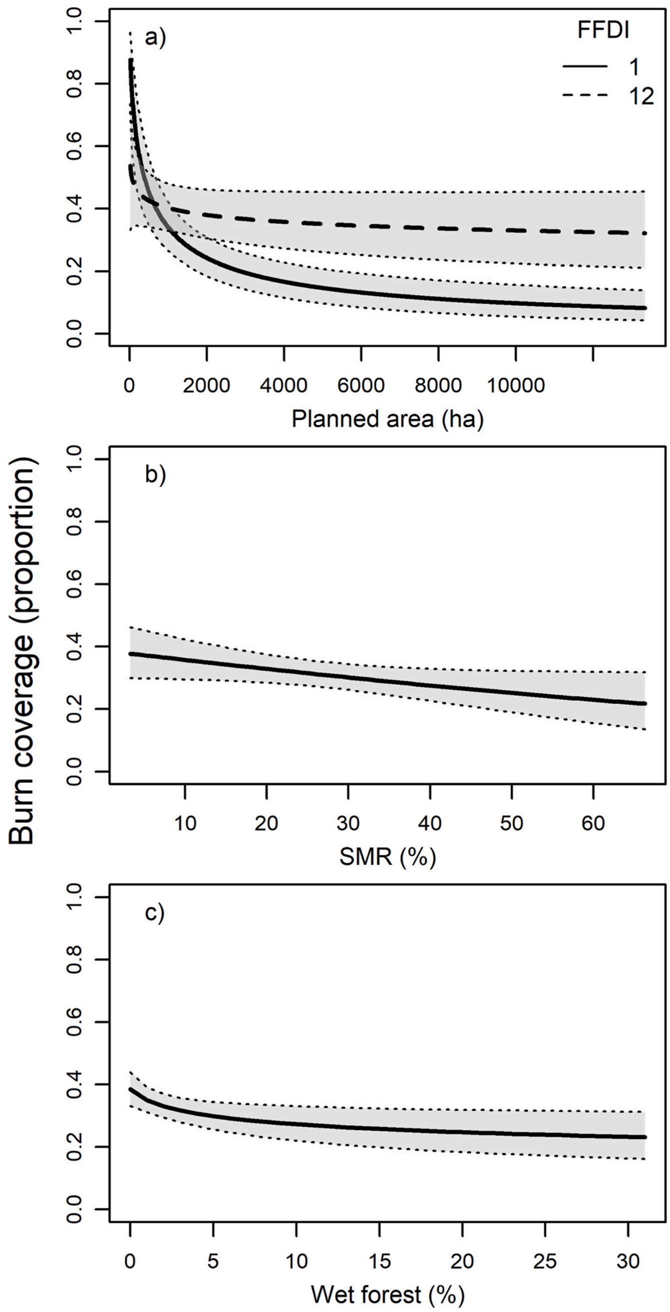

Fig. 3. Model predictions showing the response of burn coverage to (a) planned burn area and forest fire danger index (FFDI), (b) root zone soil moisture (SMR), and (c) the extent (%) of wet forest (WF) within the burn extent. Variables not shown in the plots have been held constant at their mean value. Bold lines show the predicted median and polygons show the 95% credible intervals.

to assess the effectiveness of fuel reduction burning for moderation of wildfire severity in forests of south-eastern Australia, while accounting for the extent of treated fuels within the burn extent (i.e., burn coverage). The effect of fuel reduction burning on wildfire severity was dependent on fire weather when assessed at the point scale. Wildfire severity increased with increasing time since fuel reduction burning and severity of fire weather across the targeted vegetation communities. Consequently, the greatest likelihood of high severity canopy defoliating fires (i.e., HCS, CB) occurred in fuels older than 5 years under severe weather (FFDI ≥50). Fuels strongly limited wildfire occurrence and severity within approximately 1–2 years of fuel treatment, after which fire weather became an increasingly dominant driver of severity. At the scale of the 250-ha landscape, increasing the extent of recently (<5 years) treated fuels marginally decreased the degree of canopy scorch, but had little effect on the extent of unburnt vegetation or canopy fire. The high level of uncertainty around the effect of fuel treatment on fire severity patterns within landscapes likely stems from the spatial heterogeneity of other influences on fire severity (e.g., vegetation type, topography). These findings suggest that during extreme fire events, fuel reduction burning is effective at aiding wildfire suppression locally (i.e., near assets), but the effect on the overall extent and severity of wildfire, and ultimately the exposure of assets, at landscape and bioregional scales is highly variable (e.g., Clarke et al., 2022; Collins et al., 2022).

Fuel reduction burning resulted in a clear reduction in wildfire severity across the forest and shrubland communities examined, despite extreme and prolonged drought prior to the 2018-19 and 2019-20 fire seasons. Our results indicate that fuel reduction burning moderated wildfire severity within the first ~3–8 years of treatment in forests, and within ~3–5 years of treatment in shrublands. Tolhurst and McCarthy (2016) found similar results in their assessment of the 2003 Alpine mega-fire in south-eastern Australia, which burnt over 1 million ha under extreme drought conditions: fuel reduction burning had the greatest effect on wildfire severity within 3 years of treatment, with little effect beyond 10 years. Studies examining the effect of time since fire, irrespective of fire type, on wildfire severity in eucalypt forests have shown moderation of wildfire severity up to ~10–15 years in some cases (e.g., Bradstock et al., 2010; Price and Bradstock, 2012). The longer effect of past burning reported in these studies may be attributable to the inclusion of wildfires in the fuel age assessment (i.e., wildfires consume more fuel than fuel reduction burns) (Nolan et al., 2021) or because of less severe antecedent drought (e.g., Collins et al., 2019).

Fire weather was the dominant control of fire severity under conditions that were not strongly fuel limited (i.e., TSFR >3 years), and ultimately determined extent of fire severity classes at the local landscape scale. Wildfire severity shifts from predominantly patchy and/or low severity understorey fires when FFDI is low to high (FFDI <25), to predominantly high severity canopy defoliating fires (i.e., HCS, CB) when fire weather is severe or extreme (FFDI ≥50). The FFDI value at which wildfires are predominantly canopy defoliating varies across vegetation types, increasing from the short statured shrublands (FFDI ≈ 40) to open-forests (FFDI ≈ 60) to tall-open forests (FFDI ≈ 80), reflecting differences in ecosystem productivity and canopy height and hence the susceptibility of crowns to scorch and consumption (Burrows, 1997; Zylstra et al., 2016). More severe fire weather is necessary to cause canopy scorch or consumption in tall-open forests compared to open-forests, as tall-open forest occurs in areas of high precipitation or moist sheltered topographic locations (Cheal, 2010; Gill and Catling, 2002), that are inherently less flammable than open-forests owing to greater fuel moisture content (Collins et al., 2019; Nolan et al., 2016). Consequently, greater energy is required to heat fuels to their ignition temperature in tall-open forests (Nolan et al., 2020; Zylstra et al., 2016).

Variables related to ecosystem water availability (i.e., CFM, SMR) did not impose strong control on fire severity patterns in our point-scale analysis, which was unexpected given the influence of fuel moisture on large wildfire occurrence across the study region (Bradstock, 2010; Nolan et al., 2016). It is likely that water availability was not influential owing to the narrow range of soil and canopy foliar moisture conditions sampled in our study, which was highly skewed towards low values (i.e., SMR <15%, CFM <100%). Much of south-eastern Australia was in the midst of an extreme multi-year drought during the 2018-19 and 2019-20 fire seasons, with a strong water deficit occurring across the study area (Abram et al., 2021). Drought conditions recorded across both fire seasons surpassed critical flammability thresholds (see Collins et al., 2022), with canopy foliar moisture content falling below reported thresholds for large fire occurrence (i.e., 100% moisture content; Nolan et al., 2016) across much of the study area (Fig. S3). Although root zone soil moisture and canopy foliar moisture appeared to influence fire severity patterns at the landscape scale, this is likely due to spatial

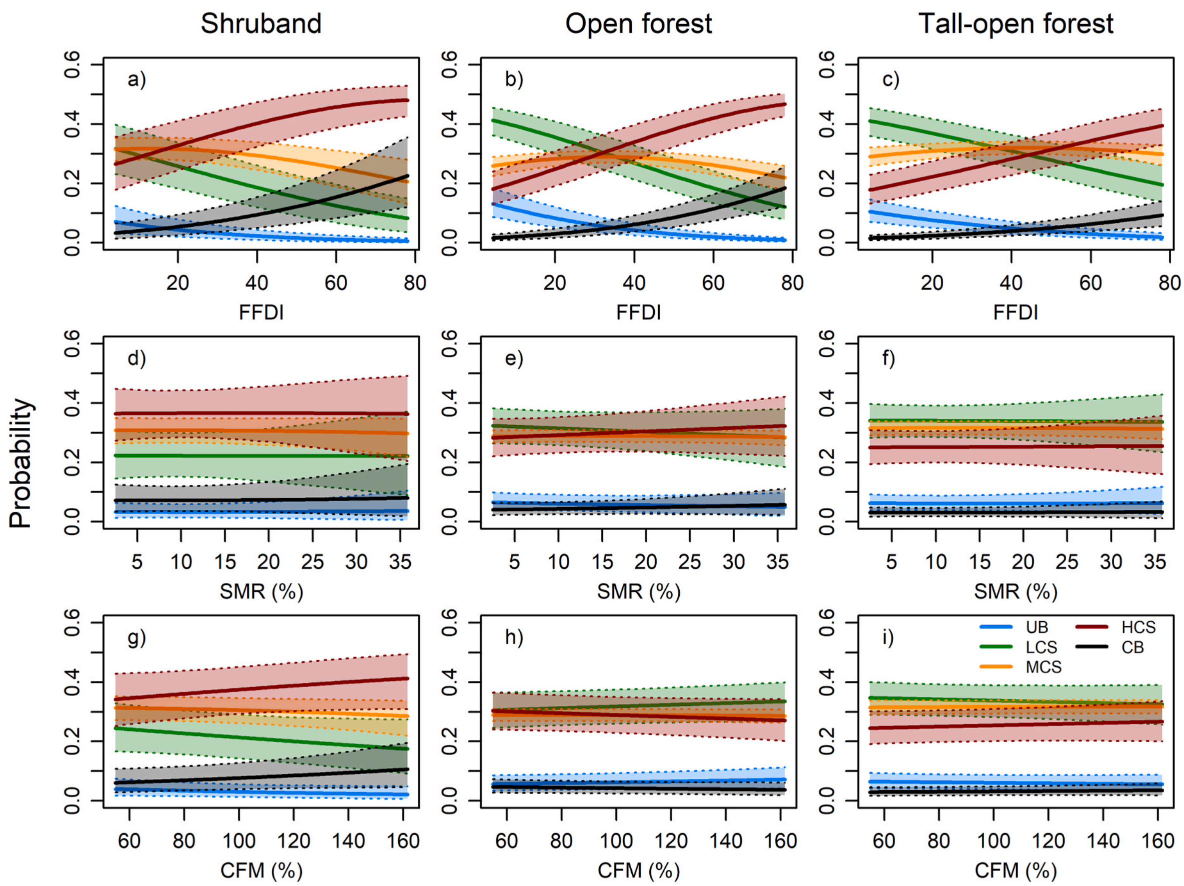

Fig. 4. The modelled response of the probability of each wildfire severity class to fire weather (FFDI; a - c), root zone soil moisture (SMR; d - f) and canopy foliar moisture (CFM, g - i). The left column is the prediction for shrublands (SH), the middle column is for open-forest (OF) and the right column is for tall-open forest (TOF). The coloured lines represent the different severity levels: unburnt (UB, blue), low canopy scorch (LCS, green), moderate canopy scorch (MCS, orange), high canopy scorch (HCS, red), and canopy burnt (CB, black). The bold lines show the predicted median, and the polygons show the 95% credible intervals. Predictions were made for areas not treated with fuel reduction burning. Variables not shown in the plots were held constant at their mean value.

patterns in water availability and their influence on vegetation structure and fuel moisture (i.e., less flammable tall-open forest, ash forest and rainforest occupy areas with greater water availability) (Gill and Catling, 2002).

Fuel reduction burns conducted in the decade prior to the 2019-20 Black Summer fires displayed considerable patchiness, with an average burn coverage of ~40% and a wide range across burns. Notably, burn coverage was considerably lower than the desired targets for APZs and BMZs, where moderation of wildfire behaviour is the primary objective. The low coverage of burns in the APZ and BMZ was likely due to a combination of factors, including weather and fuel availability constraints, and the challenges of achieving high burn coverage near assets and in complex terrain (Duff et al., 2019; Penman et al., 2007). For example, burns conducted in APZs and BMZs pose a major hazard to human life and property if they escape containment lines and, therefore, are almost always conducted under low to moderate FFDI (i.e., <12). Resourcing issues coupled with a narrow burn window can also necessitate burning under conditions that are less-than-ideal for achieving high burn coverage. Additionally, burn coverage objectives in these zones can sometimes vary on a case-by-case basis due to various factors including but not limited to ecological or practical considerations. In the LMZ, where there is a focus on balancing fuel reduction and ecological objectives, burn coverage was typically within the desired range (i.e., <50% cover).

Our assessment of the determinants of fuel reduction burn coverage emphasises that there are diminishing returns with increasing planned extent of burns. The tendency for large burns (e.g., >1000 ha) to have lower burn coverage than smaller burns (e.g., <500 ha) is consistent with the limited published data available (Duff et al., 2019; McCarthy et al., 2017; Penman et al., 2007). The heterogeneity of landscape characteristics affecting stand flammability (e.g., topographic position, riparian areas, forest type) (Penman et al., 2007) and the distance between ignition lines or points increases as the planned extent of burns gets larger, resulting in high incidence of unburnt vegetation (Duff et al., 2019; McCaw and Burrows, 2020). Furthermore, burns in the LMZ will intentionally aim for low burn coverage, and very large burns (>5000 ha) typically only occur in this fuel management zone. Burning under 'high' fire weather conditions (e.g., FFDI ≥12 vs FFDI = 1) would help achieve fuel reduction objectives for larger burns in APZs and BMZs, by reducing litter bed moisture and decreasing the likelihood of fires self-extinguishing, though this comes with an increased risk of burns escaping containment lines (Luke and McArthur, 1978).

5.1. Management implications

Short-interval (i.e., 5–8 years) burns with high coverage (i.e., 80%–100% burn coverage) are prescribed to achieve localised protection of life and property in eastern Victoria, Australia. Our point-based assessment suggests that burning intervals within the range of 5–8 years will be effective for localised reduction of wildfire severity (relative to untreated fuels) in eucalypt forests even under severe fire weather and drought, the conditions when the majority of house loss and fatalities have historically occurred (Blanchi et al., 2010, 2014). However, large areas of untreated fuels present within burns (including APZs where higher burn coverage is sought) will present a hazard to assets at the wildland-urban interface (Davim et al., 2021). The inherent patchiness

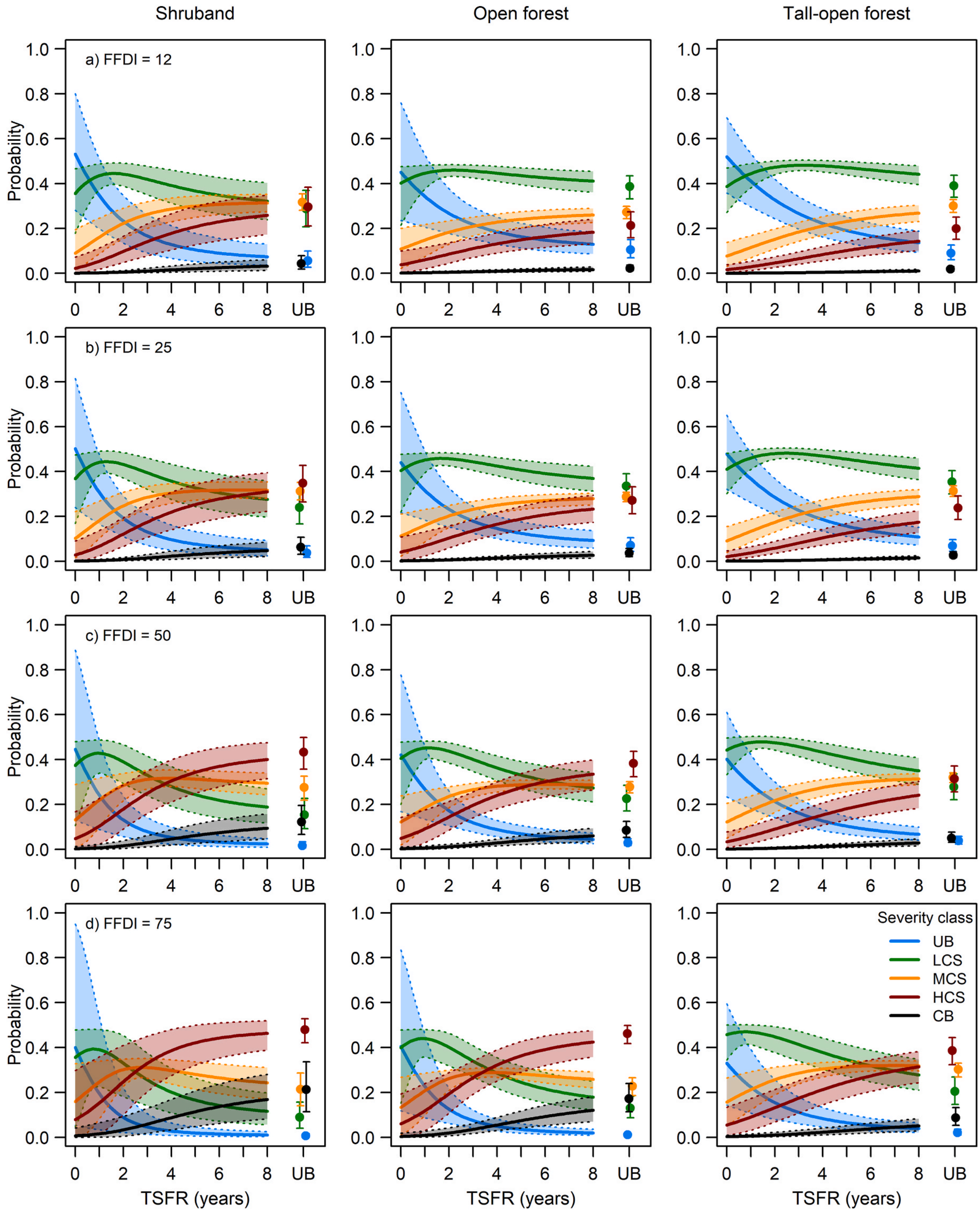

Fig. 5. Model predictions for the effect of time since fuel reduction burning (TSFR) on wildfire severity across vegetation communities and different levels of fire weather (FFDI). Values of FFDI used in the model predictions increase from the top row (FFDI = 12) to the bottom row (FFDI = 75). The left columns are predictions for shrublands (SH), the middle column is for open-forest (OF) and the right column is for tall-open forest (TOF). Fire severity levels are unburnt (UB), low canopy scorch (LCS), moderate canopy scorch (MCS), high canopy scorch (HCS), and canopy burnt (CB). The bold lines show the predicted median, and the polygons show the 95% credible intervals. Variables not shown in the plots were held constant at their mean value.

of fuel reduction burns emphasises the need for routine mapping of burn coverage to reliably quantify hazard reduction and adapt burn programs as necessary (e.g., follow up treatment of unburnt areas) (Gippsland Strategic Bushfire Management Planning Working Group, 2020). Such assessments will be aided through recent advances in prescribed burn mapping using remote sensing (e.g., Collins et al., 2020).

Results from our local landscape scale analysis suggest that broader scale application of fuel reduction burning, such as those proposed following the devastating Black Saturday wildfires in 2009 (i.e., 5% of the landscape treated per annum), would only reduce the extent of high severity canopy defoliating fire during extreme fire seasons by ~5% under ideal weather conditions (i.e., increasing RFRB from 0 to 25% reduces HCS by ~5%; Fig. 8). These marginal gains are unlikely to be realised during extreme wildfire events when large areas burn under

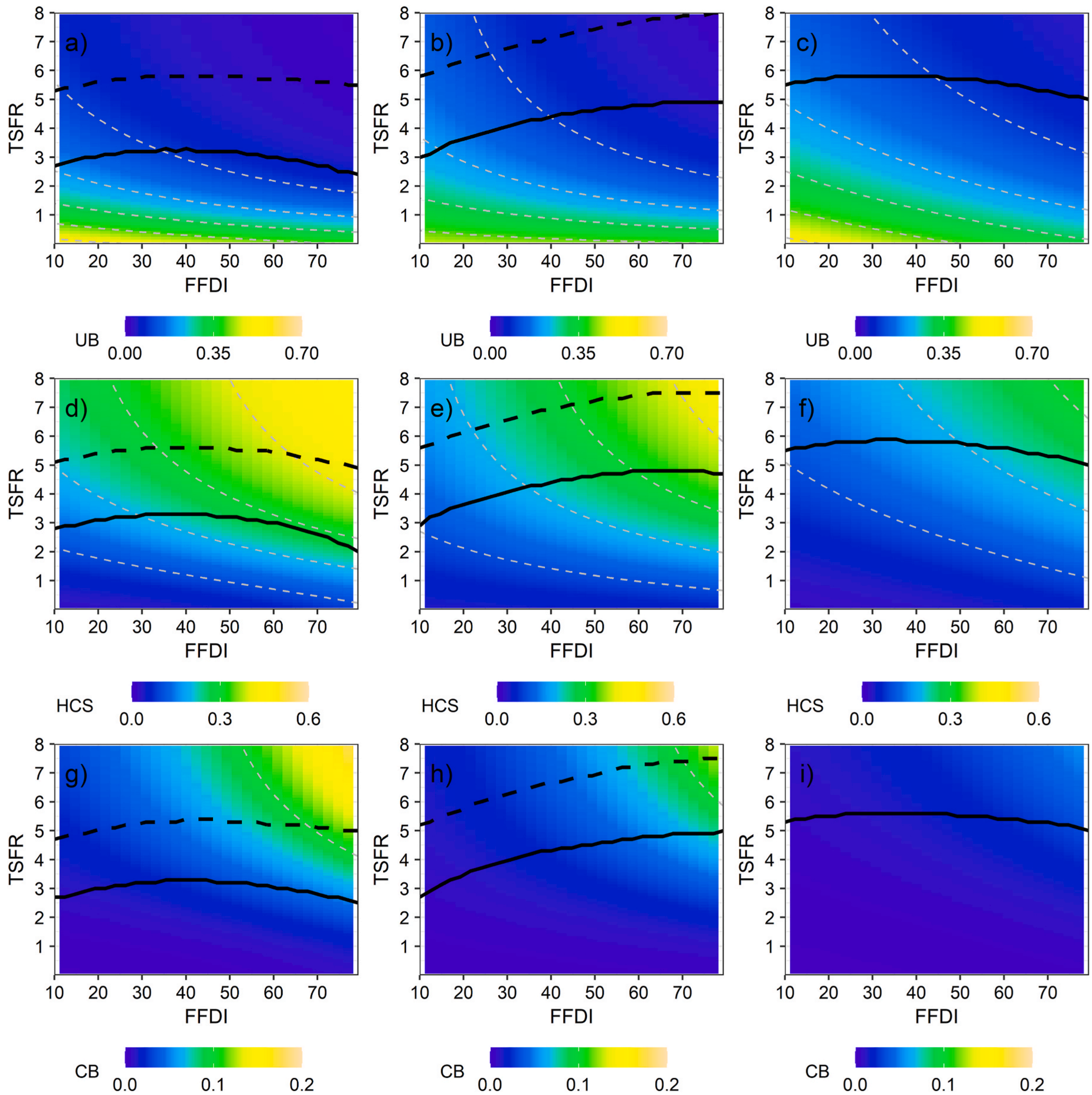

Fig. 6. The effect of time since fuel reduction burning (TSFR) and fire weather (FFDI, forest fire danger index) on the probability of three wildfire severity classes: unburnt vegetation (UB, a - c), high canopy scorch (HCS, d - f) and canopy burnt (CB, g - i). Columns are for shrubland (left; a, d, g), open-forest (middle; b, e, h) and tall-open forest (right; c, f, i). The solid black line shows the minimum time and the broken line the maximum time over which fuel reduction burning increases (UB) or decreases (HCS, CB) the likelihood of a severity class relative to unburnt areas. Heat maps show the probability of a severity class occurring and the grey broken lines are probability contours (increments = 0.1). Variables not shown in the plots were held constant at their mean value.

severe fire weather (e.g., ~50% of the 2003 Alpine fire extent; Tolhurst and McCarthy, 2016) or when pyroconvective fires occur (e.g., ~0.5 M ha in south-eastern Australia in 2019–20; Peterson et al., 2021). In these cases, fuel reduction burning is at its least effective for moderating wildfire severity. Ambitious fuel reduction programs that treat large areas of the forest landscape per annum are likely to be ineffective in reducing high severity canopy damage within the extent of an intense, large-scale wildfire occurring during an extreme fire season. These conclusions are supported by a recent simulation study that found increased fuel reduction effort (i.e., from 1% to 15% of the landscape) could not offset the elevated wildfire risk created by the extreme weather conditions experienced during the 2019-20 Black Summer (Clarke et al., 2022).

Extreme wildfire events burning under severe fire weather and drought have increased the average contribution of high severity fire (i.e., HCS and CB) to annual area burned by ~10–20% over the past three decades across the temperate forest biome in south-eastern Australia (Collins et al., 2022). Increasing the extent and frequency of fuel reduction burning in isolation will do little to offset the impacts of extreme fire seasons, such as the 2019-20 Black Summer (see Clarke et al., 2022), on forest ecosystems under a changing climate. Though, fuel reduction burning will have benefits in terms of reducing wildfire impacts during more moderate fire seasons. To cope with the anticipated increases in wildfire extent and severity, there is an urgent need to identify a complimentary suite of management actions that increase ecosystem resistance and resilience to high severity fire at the landscape-scale. These may include transforming fuel structure and associated vegetation at landscape scales using a myriad of treatments such as re-establishment of traditional burning regimes, stand improvement (e.g., thinning), and mechanical fuel reduction, which would complement fuel reduction burning programs aimed at reducing risk to communities and biodiversity. However, such changes will require a much greater shift in management than previously permitted to combat these novel fire regimes (see Bowman et al., 2020a; Fletcher

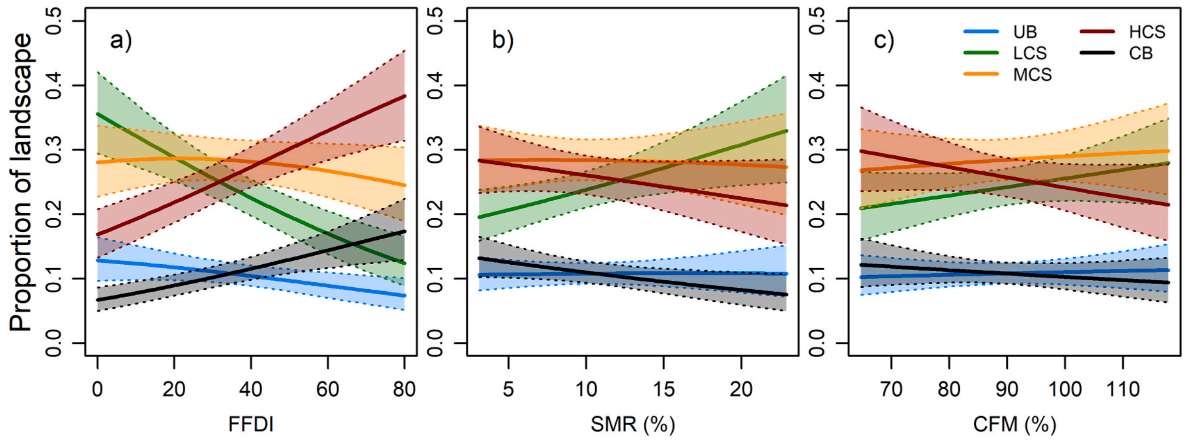

Fig. 7. Changes in the proportion of each fire severity class within the 250 ha landscapes in response to a) forest fire danger index (FFDI), b) soil moisture content in the root zone (SMR) and c) canopy foliar moisture (CFM). Fire severity classes are unburnt (UB), low canopy scorch (LCS), moderate canopy scorch (MCS), high canopy scorch (HCS), and canopy burnt (CB). The bold lines show the predicted median, and the polygons show the 95% credible intervals. Variables not presented in a plot were held constant at the mean.

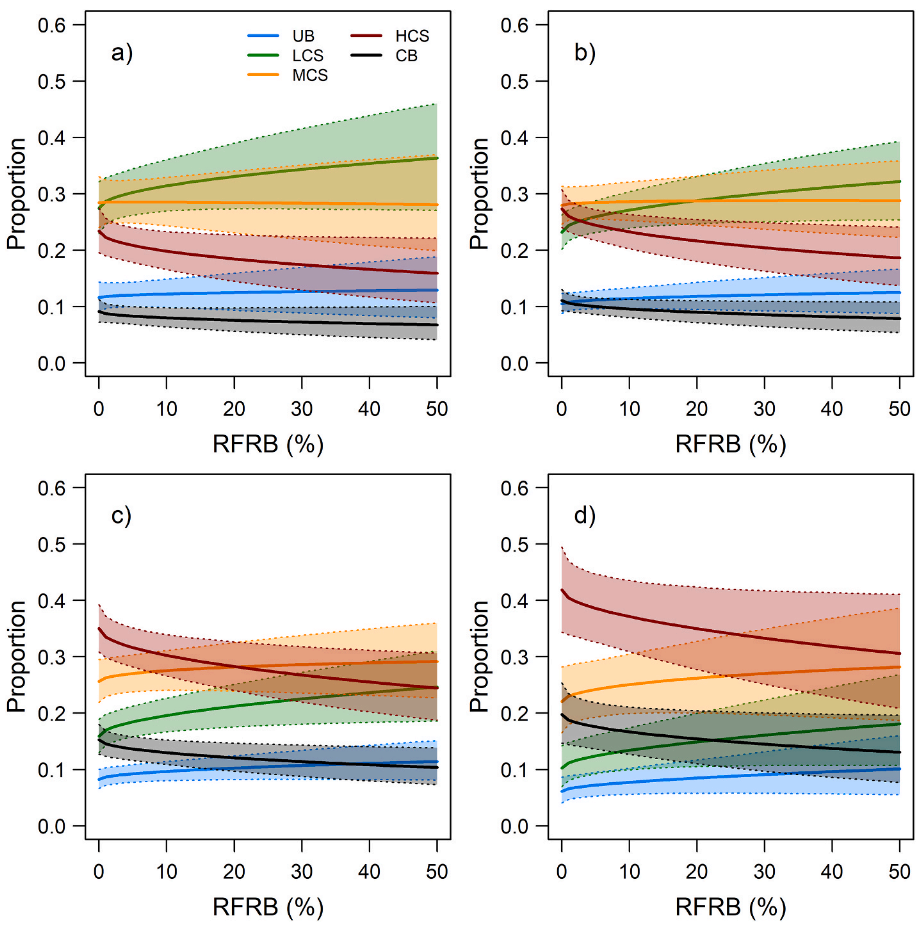

Fig. 8. Changes in the proportion of each fire severity class in response to the amount of recent fuel reduction burning (RFRB) within the 250 ha landscape. The panels show relationships for four levels of FFDI: a) FFDI = 12; b) FFDI = 25; c) FFDI = 50, and; d) FFDI = 75. Fire severity classes are unburnt (UB), low canopy scorch (LCS), moderate canopy scorch (MCS), high canopy scorch (HCS), and canopy burnt (CB). The bold lines show the predicted median, and the polygons show the 95% credible intervals. We limited RFRB to less than 50%, as relatively few data points exceeded this value (n = 12). Variables not presented in a plot were held constant at the mean.

et al., 2021; Stephens et al., 2020).

5.2. Conclusions

Our results show that FRB is effective for the localised reduction of wildfire severity during extreme wildfire events in temperate woody ecosystems of south-eastern Australia. Areas recently treated with FRB (e.g., within 3 years) will be most effective for asset protection, though considerable residual fuel hazard will often remain within the extent of fuel reduction burns owing to their patchy coverage. At the landscape scale, increased application of FRB can yield modest reductions to the degree of crown scorch. Our findings suggest that increased FRB alone will be insufficient to mitigate the impacts of worsening fire weather conditions on temperate woody ecosystems in south-eastern Australia.

CRediT authorship contribution statement

Luke Collins: Conceptualization, Data curation, Formal analysis, Methodology, Visualization, Writing - original draft. Raphael Trouvé: Conceptualization, Formal analysis, Methodology, Visualization, Writing - original draft. Patrick J. Baker: Conceptualization, Writing - review & editing. Brett Cirulus: Data curation, Writing - review & editing. Craig R. Nitschke: Conceptualization, Writing - review & editing. Rachael Nolan: Conceptualization, Writing - review & editing. Luke Smith: Data curation, Writing - review & editing. Trent D. Penman: Conceptualization, Writing - review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

Acknowledgements

We thank Rob Poore from the Department of Energy, Environment and Climate Action for providing comments on an earlier version of the manuscript. This research did not receive any specific grant from funding agencies in the public, commercial, or not-for-profit sectors.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2023.118171.

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