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Flammability dynamics in the Australian Alps

PHILIP JOHN ZYLSTRA*
School of Biological Sciences, University of Wollongong, Northfields Ave, Wollongong, New South Wales, Australia (Email: pzylstra@uow.edu.au)

Abstract Forests of the Australian Alps (SE Australia) are considered some of the most vulnerable to climate change in the country, with ecosystem collapse considered likely for some due to frequent fire. It is not yet known, however, whether increasing fire frequency may stabilize due to reductions in flammability related to reduced time for fuel accumulation, show no trend, or increase due to positive feedbacks related to vegetation changes. To determine what these trends have been historically, dynamics were measured for 58 years of mapped fire history. The 1.4 million ha forested area was divided into broad formations based on structure and dominant canopy trees, and dynamics were measured for each using flammability ratio, a modification of probability of ignition at a point. Crown fire likelihood was measured for each formation, based on satellite-derived measurements of the 2003 fire effects across a large part of the area. Contrary to popular perception but consistent with mechanistic expectations, all forests exhibited pronounced positive feedbacks. The strongest response was observed in tall, wet forests dominated by Ash-type eucalypts, where, despite a short period of low flammability following fire, post-disturbance stands have been more than eight times as likely to burn than have mature stands. The weakest feedbacks occurred in open forest, although post-disturbance forests were still 1.5 times as likely to burn as mature forests. Apart from low, dry open woodland where there was insufficient data to detect a trend, all forests were most likely to experience crown fire during their period of regeneration. The implications of this are significant for the Alps, as increasing fire frequency has the potential to accelerate by producing an increasingly flammable landscape. These effects may be semi-permanent in tall, wet forest, where frequent fire promotes ecosystem collapse into either the more flammable open forest formation, or to heathland.

Key words: Australian Alps, climate change impacts, ecosystem collapse, flammability.

INTRODUCTION

The fire season is both lengthening and becoming more severe in Australia, in line with global trends (Clarke *et al.* 2013; Flannigan *et al.* 2013; Jolly *et al.* 2015). Coupled with the increase in lightning activity (Reeve & Toumi 1999), this has the potential to greatly increase the impact of fire on forest and other ecosystems (Krause *et al.* 2014), with consequent feedbacks into the carbon cycle (e.g. (Fisher *et al.* 2016).

The corollary to external climatic pressure is the internal flammability response of the vegetation. If recently burnt, post-disturbance forests are less flammable, then the feedback is negative and stability is restored. If, however, the feedback is positive and post-disturbance forests are more flammable, then the influence of these external climatic drivers will be amplified by the internal dynamics of the forest (Bowman *et al.* 2014b). Research into these feedbacks has become a priority field in our understanding of the earth system (Bowman *et al.* 2009; Harris *et al.* 2016).

*Corresponding author.
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Flammability feedbacks

The inherent flammability of an ecosystem has the potential to change as its components vary over time through growth, succession and production–decomposition processes acting on energy storage. In the immediate period following a fire, the forest is partially cleared, bare soil produced and the capacity for fire to spread is low and increasing. In the fuel-age paradigm that underpins much fire management and risk modelling (Zedler & Seiger 2000; Fernandes & Botelho 2003; McCaw 2013), this rising trend is assumed to continue towards a point of equilibrium, consistent with the dynamics of fuel accumulation (Olsen 1963). As a result, the expectation is that forests beyond a certain age are always more flammable than young forests. In many instances where flammability dynamics have been measured over longer time periods, however, the initial increase in flammability has been a temporary trend, and flammability has been seen to decrease again after the forest has been re-established (e.g. Cochrane *et al.* 1999; Kitzberger *et al.* 2012; Perry *et al.* 2014; Tepley *et al.* 2016). This scenario was proposed in the evolutionary theory of the 'angiosperm revolutions' by (Bond & Midgley 2012), where angiosperm forests gained competitive advantage through their ability to create

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mature environments that suppressed the spread of fire. The causal mechanisms for this have been examined and described at length for given examples (Kitzberger *et al.* 2012, 2016), and related to changes in forest structure and species composition (Zylstra 2011; Zylstra *et al.* 2016) through the course of post-disturbance succession (Zylstra 2013, 2016).

In this study then, flammability dynamics are described in a three-stage model, consistent with the Chapman–Richards conception of forest growth (Fekedulegn *et al.* 1999). This commences with two post-disturbance phases comprising a 'young' phase immediately following fire, where some or all of the forest has been physically cleared. This is followed by a 'regrowth' phase in which re-establishment occurs and flammability changes rapidly, and culminates in a quasi-equilibrium 'mature' phase. Positive and negative feedbacks are defined by the difference in average flammability between mature forest and the two post-disturbance phases, so that more flammable mature stands define negative feedbacks, and less flammable mature stands define positive feedbacks.

Defining flammability

Assessing this requires a clear definition of flammability. The term refers to the likelihood or ability of something to burn (Gill & Zylstra 2005; Stevenson 2010), and is physically derived from the components ignitability, combustibility and sustainability (Philpot 1970). These can operate independently as countermeasures, or in ways that amplify each other (Gill & Zylstra 2005). At the level of an individual fuel particle such as a leaf, it is possible to measure each component separately and precisely (Pérez-Harguindeguy *et al.* 2013), but combinations of these traits into a single statistic can be somewhat arbitrary without a methodology to weigh their relative effects on an objective outcome. As a result, flammability is often better defined with three separate components (Pausas *et al.* 2017).

This situation changes as individual fuel components are aggregated into fuel beds, plants or communities of plants. The interaction between each component forms a complex system, where the outcome is predicted not by a simple combination of the inputs (de Magalhães & Schwilk 2012; Grootemaat *et al.* 2017; Wyse *et al.* 2017), but emerges from numerous feedbacks and dynamic changes (Zylstra 2011; Zylstra *et al.* 2016). Flame length for example corresponds to combustibility, but the flame from a plant is an emergent property that is also determined by the ways that leaf ignitability and sustainability affect how many leaves are burning simultaneously (Zylstra *et al.* 2016). When viewed as a complex system with fire behaviour as an emergent property, a single metric of flammability is possible at such larger measurement scales. To be meaningful in an ecological or evolutionary sense, such metrics may still require additional description (Pausas *et al.* 2017).

Two requirements are therefore necessary for an adequate measure of flammability as a single, unified statistic:

1. It must emerge naturally from the three components of ignitability, combustibility and sustainability
2. It must define the likelihood or ability of burning, as per the broad definition"

The metrics of probability of ignition at a point (Gill *et al.* 2000) and severity both satisfy these criteria. Probability of ignition at a point is a measure of fire frequency analogous to ignitability (Gill & Zylstra 2005), but it emerges from the interactions of ignitability (likelihood of fire occurrence, rates of spread), combustibility (flame dimensions affecting suppression effectiveness and spread via spotting) and sustainability (the length of time that a fire is burning and able to spread). Together, these define the likelihood of burning in the horizontal dimensions. The vertical dimension can be measured by severity, or by the height to which vegetation is consumed by the fire at a point (Keeley 2009). This also is an effect of flame dimensions, and as described earlier, these are emergent properties of the interaction between all flammability components within a plant or stand.

Both of these statistics can be said to give contextual, rather than absolute flammability. Absolute values of flammability describe the endogenous contribution of a leaf, plant or community in units of measurement, applicable to standardized exogenous conditions. This can be measured for smaller units such as leaves or plants that can be tested under experimental conditions, but plant communities occur within a context. Exogenous drivers include the frequency of ignitions, fire contagion from adjacent communities, local weather and terrain and suppression activity. These not only influence the historic likelihood of burning, but also contribute to the conditions that have shaped the community itself.

To illustrate, rainforest often occurs in moist, sheltered areas. Shelter and moisture, however, also affect the likelihood of fire, so that the flammability of the rainforest is affected by its context and is not entirely endogenous. In contrast, absolute flammability is not limited by the actual context, but defines the flammability within a pre-determined set of conditions that may or may not be experienced by the ecosystem. Bushfire threat levels in New South Wales are for example assessed against standardized weather conditions, whether the community is desert, rainforest or alpine (NSW Rural Fire Service 2006).

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Contextual flammability can therefore be measured empirically and has value for understanding the existing situation, whereas absolute flammability must be modelled for a changed context, such as under projected climate change (Cuddington *et al.* 2013).

Flammability in the Australian Alps

The Australian Alps are vulnerable to numerous changes, and as an 'elevationally restricted mountain ecosystem' with high numbers of endemic species (Pickering *et al.* 2004), they fall into the category of first concern for Australian ecosystems at risk from climate change (Laurance *et al.* 2011). Consistent with increases in local fire weather severity (Clarke *et al.* 2013), fire frequency in the Alps has increased sharply in the 21st century (Fairman *et al.* 2016).

Positive flammability feedbacks have long been considered typical for the area by graziers (Wakefield 1970; Zylstra 2006) and naturalists (Helms 1896; Stretton 1939; Fawcett 1947), although the evidence for this was largely anecdotal until recently demonstrated for some montane (Taylor *et al.* 2014), subalpine (Zylstra 2013) and alpine (Camac *et al.* 2017) communities. In the context of external factors increasing fire frequency, such amplifying feedbacks have the potential to produce 'landscape traps' (Lindenmayer *et al.* 2011), where fire feeds fire until tipping points are reached and ecosystem collapse occurs in vulnerable communities (Kitzberger *et al.* 2012).

In the montane zone, Bowman *et al.* (2014a) have measured localized ecosystem collapses produced by fire in tall, obligate-seeding alpine ash (*Eucalyptus delegatensis* subsp. *delegatensis*, Myrtaceae) forests. Although the likelihood of total collapse has so far been assessed for only the *E. regnans* (Myrtaceae) component of montane forests (Ferguson 2010; Bowman *et al.* 2014a), there is significant risk of it for some alpine communities such as snow patch herbfield or feldmark (Green & Pickering 2009; Williams *et al.* 2015), and long-term changes have been measured across sub-alpine forests and woodlands (Fairman *et al.* 2017). Fire can also act as a catalyst for climate-driven change along ecotones between formations, as observed in Yellowstone National Park (Donato *et al.* 2016).

Such concerns may for some be counterintuitive in Australian eucalypt forests, which are frequently described as 'fire adapted', with arguments for deliberate management increases in fire frequency coming from both political and some academic sources (Teague *et al.* 2010; Attiwill & Adams 2013; Baker & Catterall 2016). While debate continues over the ecological veracity of this characterization (Bradshaw *et al.* 2011), care is needed that ecological requirements do not become conflated with flammability dynamics.

Aims

In order to better define the risk posed to the Australian Alps through fire, the aim of this paper is to quantify the fire-flammability feedbacks across all broad forest formations of the Australian Alps.

MATERIALS AND METHODS

Location

The study was conducted in the forested part of the Australian Alps interim bioregion (Hutchinson *et al.* 2005) that is contained within the Australian Alps National Parks. As detailed below, these forests cover 1.4M ha across the states of New South Wales and Victoria, along with the Australian Capital Territory.

The environment is diverse, largely due to topography. Elevation ranges from close to sea level in the south east, to the highest forested areas in Australia at 1960 m.a.s.l., and the area spans two temperate climatic divisions with distinctly dry summers ranging from mild to hot (Stern *et al.* 2000). Mean annual rainfall ranges from 500 mm to over 2300 mm, and snow can persist for up to 6 months in areas above 1400 m.a.s.l. (Clayton-Greene & Ashton 1990; NSW Office of Environment and Heritage 2016). Pre-European fire return intervals ranged from multiple decades to centuries (Zylstra 2006; Mooney *et al.* 2011).

Forest ecosystems

Forests were broadly grouped in formations divided by structure (Specht 1970) and dominant tree species, combining units from Victorian Ecological Vegetation Groups (Department of Sustainability and Environment 2010) and NSW/ACT Formations (Gellie 2005). These were tall wet forest, open forest, subalpine open forest-woodland, dry open forest and low, dry open woodland (Table 1, Fig. 1).

Fire history

The primary data used for all fires mapped for 58 years from 1957 to 2015 were records sourced from State Government agencies (New South Wales Office of Environment and Heritage, Australian Capital Territory Emergency Services Agency, and the Victorian Department of Environment and Primary Industries, unpublished data). These were checked for duplicate polygons, and fires that had not been entered from the archived paper records were added to the dataset from that used by (Zylstra 2013). Some fires had been mapped prior to this period, however, the quality and frequency of this was inconsistent, rendering the data too poor for analysis.

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Table 1. Forest formations within the study area, showing details of the component formations† in the New South Wales and Australian Capital Territory forests, followed by the Ecological Vegetation Community (EVC) groups‡ in Victoria

Mapped area (ha)	Elevation range (m.a.s.l.)	Canopy height (m)	Typical canopy species	Formations EVC Groups	Indicative structure
Tall Wet Forest					
228888	<1400	20–50	*Eucalyptus delegatensis, E. regnans, E. dalrympleana, E. pauciflora, E. fastigata* (Myrtaceae)	Ash eucalypt forests, Rainforests Wet or damp forests, Rainforests	
Open Forest					
486891	700–1300	20–35	*Eucalyptus dalrympleana, E. robertsonii subsp. robertsonii, E. macrorhyncha, E. bridgesiana, E. pauciflora, E. viminalis, E. rubida subsp. rubida, E. aggregata, E. stellulata* (Myrtaceae)	Moist eucalypt forests, Montane tableland forests, Swamp forests/ sedgelands Montane shrublands, grasslands, or woodlands	
Subalpine Forest and Woodland					
189598	1000–1960	5–15	*Eucalyptus debeuzevillei, E. niphophila, E. pauciflora* (Myrtaceae)	Subalpine low forests Subalpine shrublands, grasslands or woodlands	
Dry Open Forest					
417005	300–1100	15–20	*E. macrorhyncha, E. rossii, E. dives, E. mannifera* (Myrtaceae)	Grass/shrub forests Dry forests	
Low, Dry Open Woodland					
94102	<550	15–30	*Eucalyptus blakelyi, E. melliodora, E. bridgesiana, E. albens, E. polyanthemos subsp. polyanthemos* (Myrtaceae), *Callitris glaucophylla* (Cupressaceae)	Grassy woodlands/ grasslands Lower slopes or hills woodlands	
Total					
1416586					

†(Gellie 2005), ‡(Department of Sustainability and Environment 2010).

Analysis 1: Flammability feedbacks

Flammability feedbacks were determined by relating time since fire to two measures of Probability of Ignition at a Point (PIP) – the probability of a cell burning, and the probability of crown fire within a burning cell. The overall feedback was measured from the first of these, whereas the second was used to provide further insight.

The required inputs for each analysis were collected in ArcGIS (ESRI 2015) from 1-ha grid cells, and analysis performed in the R Statistical Environment (R Core Team 2016).

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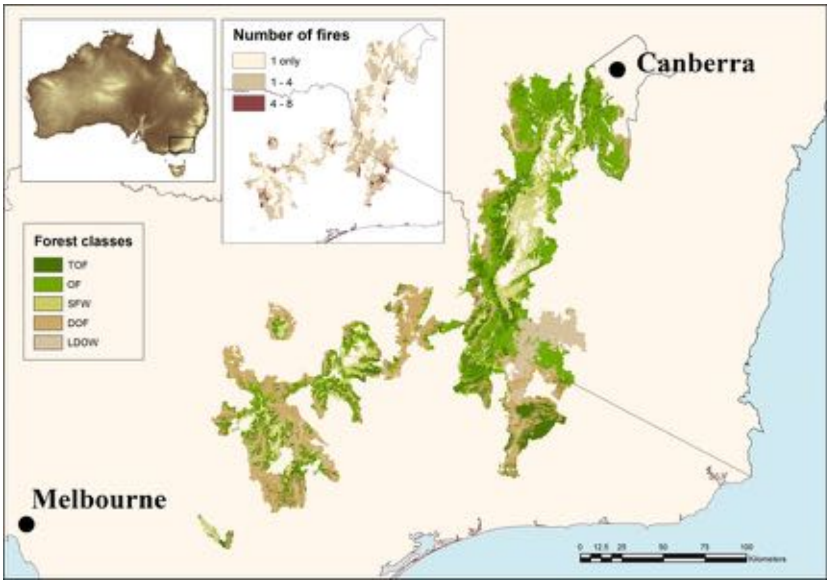

Fig. 1. Location and forest formations composing the study area in south eastern Australia.

To find the probability of a cell burning, PIP was calculated using flammability ratio FR (Zylstra 2013), as this includes a scaling factor to account for noise produced by small age-class samples. PIP is defined as the proportion of an age-class that is burnt; but smaller age-classes are likely to be either missed by a fire, or consumed entirely. To account for this noise in the data, the standard calculation (left hand term of Eqn 1) is multiplied by a scaling factor to give the area factor (*AF*, Eqn 1). Scaling is provided by the ratio of total burnt area in a given year Σ_b to the average area of each age-class $\bar{x}A$.

$$AF = \frac{A_b}{A} \cdot \frac{\bar{x}A}{\Sigma_b} \qquad (1)$$

where A_b is the area of the age-class burnt, and A is the total area of that age-class.

This can be understood by rearranging the Eqn. If the denominators are swapped, then we have the proportion of the burn footprint composed by a particular age-class (A_b/Σ_b), scaled by the relative frequency of that age-class ($\bar{x}A/A$). If all age-classes are equally likely to burn, then on average, they should make up equal parts of the burn area. Not all age-classes are equally common, however, the scaling factor in the right-hand term is required to correct this. If for example, one age-class is half as widespread as the average for the other age-classes, its presence in the burn footprint is doubled by the scaling factor to give it an equal weight to the others.

FR for a given age and year is equal to the *AF* divided by the average of all *AF* values across all ages and years, and the FR for that age across all years is the average of all FR values collected for that age.

Spatial autocorrelation is likely within a given year, as fire burns discrete patches rather than random cells. This can, however, be disregarded when multiple years are examined, as new patches are formed in each year.

As the fire history data spans multiple decades, it includes the influence of changing climates and fire suppression effectiveness. Although recently burnt forests were available for analysis in all years, the older age-classes were only available for measurement more recently, and therefore may reflect changed flammability resulting from these influences. To determine whether such an effect is present and significant, the FR was averaged for each year, for all age-classes that spanned the complete time series across all forests. This was then tested to determine whether there was a significant ($P < 0.05$) correlation between mean FR and year of burning.

Analysis 2: Likelihood of crown fire ignition

To find the likelihood of crown fire L_{cb} in any burning cell, crown fire measurements were made from difference

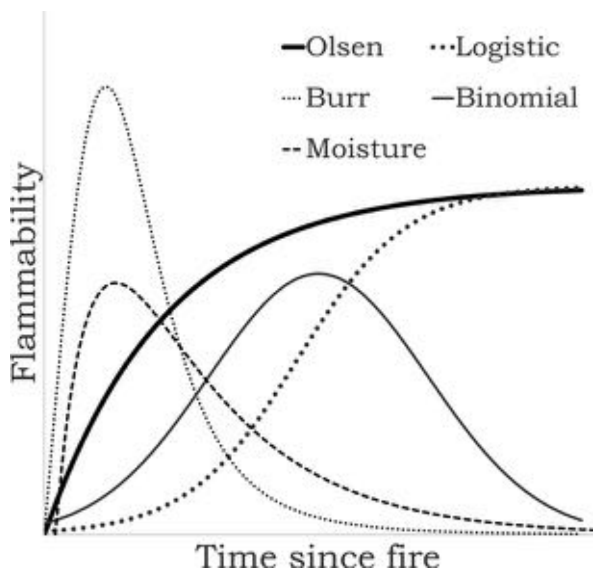

Fig. 2. Contender functions used to describe flammability dynamics, as per Eqns 2–7.

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Normalized Burn Ratio dNBR (e.g. (Keeley 2009)), mapped for a 632,448 ha subset of the 2003 bushfires in NSW and the ACT (Barrett 2006), using only the highest class of severity to denote crown fire.

As this was only measured for a single year, the dataset was smaller than the FR analysis, subject to the possibility of small, highly influential outliers and to spatial autocorrelation.

To account for noise from small outliers, these were identified using Grubbs' test (Grubbs 1950) via the 'outliers' package in R (Komsta 2011), and the largest outlier for each formation was removed if $P < 0.05$ for Grubbs' statistic. To limit the loss to data, a maximum of one outlier constituting less than 0.5% of the area of each formation was removed.

To account for spatial autocorrelation, severity measurements were made using a random grid of cells that covered 75% of each formation.

Fitting functions

To find trends in the dynamics, a series of functions were fit to the data using the non-linear least squares (NLS) package for R (Bates & Chambers 1992). Where more than one contender was identified, the best was chosen by comparing the Akaike information criterion (AIC (Akaike 1974)) using Akaike weight (Symonds & Moussalli 2011) to give the probability that this was the best approximating model. For the smaller datasets used in the second analysis, AIC_c was used (Symonds & Moussalli 2011).

All dynamics were assumed to commence with a period of low, increasing flammability, where ground fuels are absent immediately following fire. Five functions were tested for this purpose (Fig. 2). The Olsen function ((Olsen 1963), Eqn 2) and a logistic function (Eqn 3) describe negative feedbacks, however, the logistic function

Table 2. Statistics for testing hypotheses and characterizing feedbacks for the models that could be fit to the data

		Model strength			Constants			Dynamics: Max age (mean FR)			Feedback statistics	
Area burnt (ha)	Model	RSE	R^2	Akaike weight	a/sc	b/sd	Mean	Y	R	M	YE (years)	FS
Tall Wet Forest												
141,255	**Modified Burr**	**10.01**	**0.16**		**2.340*****	**1.602*****		3 (0.98)	21 (1.54)	(0.17)	0	8.3
	Olsen				1.046***	0.575						
Open Forest												
338,432	Modified Burr	15.59	0.07		2.324***	5.510*		6 (0.48)	28 (1.29)	(0.73)	4	1.5
	Binomial				129.84	35.27	−12.5					
	Olsen				1.033*	0.846						
	Burr_O	**7.39**	**0.01**		**2.221*****	**1.065*****						
Subalpine Forest & Woodland												
85,693	**Modified Burr**	**8.5**	**0.09**	**0.63**	**2.230*****	**1.163*****		6 (0.81)	25 (1.31)	(0.5)	3	2.3
	Binomial			0.37	43.319**	11.982*	17.143***					
	Olsen				1.066***	0.382						
Dry Open Forest												
331,946	**Modified Burr**	**5.85**	**0.36**	**0.80**	**2.306*****	**1.671*****		3 (0.62)	19 (1.49)	(0.5)	2	2.6
	Binomial			0.20	31.240***	6.496***	11.415***					
	Olsen				1.048***	0.631						
Low, Dry Open Woodland												
84,254	Modified Burr				2.197***	0.521		2 (0.67)	14 (1.50)	(0.67)	2	2
	Olsen				1.109**	0.403						
	Burr_O	**8.45**	**0.07**		**2.303*****	**2.285*****						

Models that could be fit using NLS are shown, regardless of their goodness of fit. The constants *a* and *b* fit either modified Burr or Olsen as indicated, and *sc*, *sd* and fit the binomial model, showing significance for each constant (*0.01, **0.001, ***0.0001). Where all constants were significant for more than one model, Akaike weight is shown for significant models to indicate the likelihood of the best approximating model. When this was >0.5, the model was further refined for some formations with the removal of outliers, in which case this adjusted model is shown with the suffix _O. Strength of the best model for each formation is given by residual standard error RSE, and R^2. The dynamics columns divide the forest into the three age-classes young Y (flammability ratio FR < 1), regrowth R (FR ≥ 1), and mature M (second period where FR < 1). The classes Y and R are described by maximum age, and all classes show mean flammability in italics. Two feedback statistics are given; these are young equivalent YE, which is the number of years after fire for which the FR ≤ the mean FR for M, and feedback strength FS, which is the mean FR for post-disturbance forests (Y and R) divided by the mean FR for M. In all cases, FS is greater than one, indicating that mean flammability of regrowing forest is greater than the mean flammability of mature forests. The best fitting model for each forest is shown in bold.

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allows for a longer period of initial reduced flammability compared to Olsen. The other functions (Burr 1942), Eqn 4 and modified to Eqn 5), the standard binomial distribution (Eqn 6) and the moisture function (Eqn 7, McCarthy *et al.* 2001) allow for positive feedbacks; with Burr capable of representing both. Equation 6 produces a very close approximation of Burr, but can be fit more easily in the R package.

$$F = a\left(1 - e^{-b.T}\right) \quad (2)$$

$$F = \frac{K}{1 + e^{(a - r.T)}} \quad (3)$$

$$F = ab\frac{0.1T^{a-1}}{\left(1 + 0.1T^{a}\right)^{b+1}} \quad (4)$$

$$F = ab\frac{0.1T^{a-1}}{\left(1 + 0.1T^{a}\right)^{b} + 1} \quad (5)$$

$$F = \frac{sc}{s\sqrt{2\pi}} \cdot e^{\frac{(-T-\bar{x})^2}{2s^2}} \quad (6)$$

$$F = a\left(1 - e^{-bT}\right)\left(c + e^{-dT}\right) \quad (7)$$

where F is a flammability metric (either FR or L_{cb}), T is years since the last fire, a to d are constants, r is the biotic

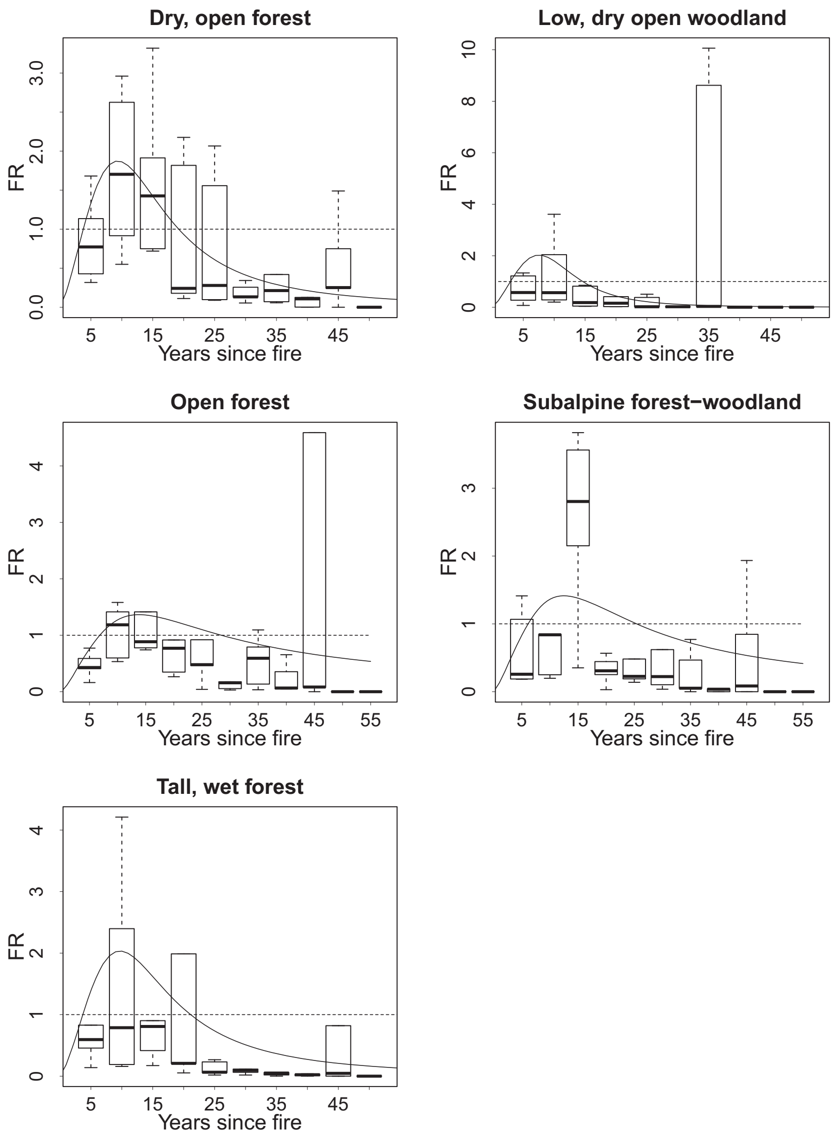

Fig. 3. Modelled flammability ratio FR (curved lines), and mean FR values grouped into 5-year clusters. The horizontal broken line shows FR = 1, so that values above the line represent ages that are more flammable than average for that community, and those below the line are less flammable than average. The formations low, dry open woodland and open forest have outliers removed. Box plots show standard quartile divisions for annual mean data, with outliers up to the default of 1.5 times the interquartile range from the box.

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rate of increase, K is an asymptote, sc is a scaling factor, s is the standard deviation and is the mean.

Models were excluded if they could not be fit using NLS, or if when fit, all constants did not have $P < 0.01$. When this occurred in the FR analysis, outliers were successively identified using Grubbs' test (Grubbs 1950; Komsta 2011) and subsequently removed before re-examination. As this dataset was much larger than that used in the second analysis, rather than set as an arbitrary limit to the size of outliers, the percentage burnt area that was removed was reported as qualifying data. The largest outlier in the dataset used for the second analysis was routinely removed, if it constituted less than 0.5% of the formation area.

A model was chosen if it achieved significance, and where more than one did so, if Akaike weight was >0.5. Only the better-fitting of either Eqns 4 or 5 was examined, as both approximate the same distribution. If no model satisfied these criteria, the null hypothesis was not rejected.

In order to best represent the data, any FR model chosen that did not achieve the highest level of significance ($P < 0.001$) was again examined for outliers. If these outliers constituted only a very small part of the burnt study area and did not change support from one alternate hypothesis to another, they were removed until maximum significance was achieved, if that was possible. Again, the area of forest removed as outliers was reported to provide qualification to these models.

Measuring feedbacks

Feedbacks were measured from the first analysis, where the null hypothesis H_0 of no feedback (i.e. flammability remains constant) was tested against the alternate hypotheses of negative feedback H_1 and positive feedback H_2. Feedbacks were positive if the average flammability of mature forests was less than the average of post-disturbance forests, negative if mature forests were more flammable, or null if no model could be fit.

Flammability dynamics in the Olsen and logistic models were divided into young forests (Y) in which modelled FR < 1, and mature forests (M) of all ages beyond this. For the other functions where FR dropped below unity at older age-classes, Y was calculated, followed by regrowth (R) where modelled FR ≥ 1, than mature forest, where FR was once again <1. The mean FR for each of these periods was measured from the raw data, after the removal of any outliers.

The strength and direction of each feedback was quantified as feedback strength (FS), which is the mean FR for post-disturbance forest (Y, or Y+R depending on the function) divided by the mean FR for M. Where the feedback was positive, young equivalent (YE) was calculated as the number of years for which the mean modelled flammability of post-disturbance forests was less than or equal to the mean flammability of mature forests.

For both analyses, the strength of the best model for each community was indicated by residual standard error for the full datasets, and by R^2. For FR, R^2 was measured against mean FR values for each age in preference to raw values, to minimize the noise arising from inter-annual stochastic differences and thereby providing a better representation of the feedback.

Final models

To find the annual likelihood of fire L_f in each age and formation, the relevant FR models were multiplied by the mean likelihood of fire L across the formation being studied (Eqn 8).

$$L = \frac{\Sigma_{bY}}{\Sigma_{AY}} \tag{8}$$

where Σ_{AY} is the sum of all areas of known age for all years, and Σ_{bY} is the sum of all burnt areas within those known ages, for all years.

To find the annual likelihood of crown fire L_c in each age and formation, the relevant models for likelihood of crown fire in a burning stand L_{cb} were multiplied by L_f.

RESULTS

Flammability feedbacks

Flammability ratio could be measured for the 52 years following 1964 in all formations except low, dry open woodland, where the first fire to burn forest of a known age was in 1965 (Appendix S1). Examination of the combined forest dataset found no evidence that FR changed due to climate or management, as there was no significant correlation

Table 3. Statistics for the models that could be fit to crown fire data (L_{cb})

			Model strength		Constants		
Formation	Model	Samples	RSE	R^2	a/sc	b/s	$\bar{x}$
Tall, wet forest	Binomial	40	0.057	0.17	1.494***	6.655***	23.704***
Open forest	Binomial	30	0.078	0.12	4.184***	11.184***	23.206***
Dry, open forest	Modified Burr	30	0.167	0.01	1.455***	0.992*	
Subalpine forest-woodland	Binomial	26	0.132	0.11	4.010***	6.63**	20.356***
Low, dry open woodland	Mean	20					0.107

Model strength is given by residual standard error RSE and R^2. The constants sc, s and describe the binomial model (Eqn 6), and asterisks indicate their significance (*0.01, **0.001, ***0.0001). Where no model could be fit, the mean of the dataset is given as the Null value.

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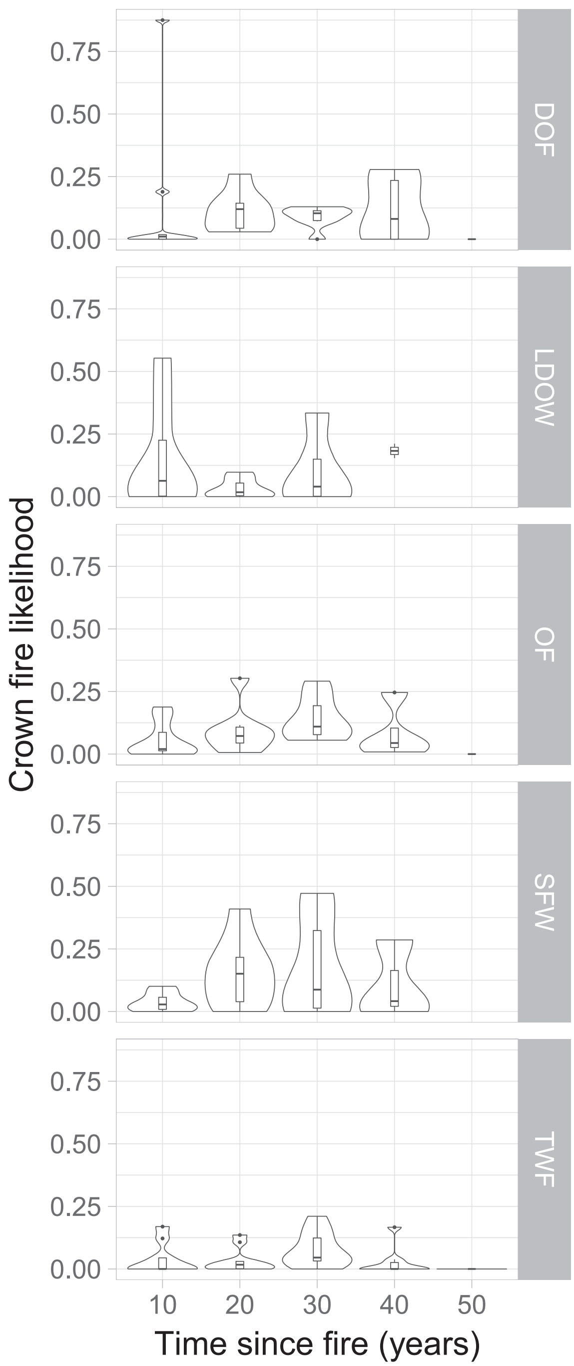

Fig. 4. Frequency of crown fire (the highest dNBR class) in decadal groupings of each formation. As per (Hintze & Nelson 1998) violin plots are composed of box plots indicating data range, quartiles and median, and the shaped area shows the density trace. Labels refer to dry, open forest (DOF), low, dry open woodland (LDOW), open forest (OF), subalpine forest and woodland (SFW), tall, wet forest (TWF).

between the year of fire and the mean FR of the ages 1–6 that were present in all years examined ($P = 0.133$).

The only significant models in all formations were Burr, modified Burr and the binomial distribution (Table 2, Fig. 3), with modified Burr providing the strongest fit in all cases. Dry, open forest, subalpine forest-woodland and tall, wet forest all had the highest category of significance for the study ($P = 0.0001$) when modelled from the full dataset using modified Burr. Open forest was significant with the full dataset ($P = 0.01$), but removal of two outliers constituting 0.007% of the studied burnt area increased this to $P = 0.0001$. Low, dry open woodland was only significant if the largest outlier was removed, but removal of another two outliers constituting 0.183% of the burnt area produced a model with $P = 0.0001$. Large outliers remained in both, but were not removed after maximum significance was reached.

Feedback strength was greater than one for all formations, indicating that flammability of post-disturbance forests was higher than that of mature forests.

Likelihood of crown fire ignition

The number of age-classes burnt in 2003 and for which data was available varied between 20 for low, dry open woodland, and 40 for tall, wet forest, allowing functions to be fit to all formations except for the woodland (Table 3, Appendix S2). The trend was binomial in three cases, and followed a modified Burr curve for dry, open forest due to a weak trend of more crown fire in the first two decades of post-fire recovery (Fig. 4). Although this model was weakly significant, it had little explanatory power ($R^2 = 0.01$).

Final models

The mean likelihood of fire per annum L varied only slightly between formations when all ages were combined (Table 4), however, the formations of open forest and subalpine forest-woodland differed notably from the others when age effects were taken into account by finding L_f (Fig. 5a). This pattern varied again when likelihood of crown fire at a burning site L_{cb} was considered (Fig. 5b), and as a result, the combined function L_c produced large differences in the annual likelihood of crown fire at a point (Fig. 5c).

DISCUSSION

Contrary to widely held perceptions of eucalypt forests, feedbacks in the study area have been pronounced and are positive in all formations over the

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past half century. Forest stands have burnt 1.5 (open forest) to 8.3 (tall wet forest) times more often in post-disturbance forest than in mature forest, and crown fires appear to have been mostly confined to post-disturbance stands.

Time since fire affected the flammability of all formations, but the proportion of its influence varied between them. The highest R^2 was measured for dry, open forest, where time since fire has contributed to more than one-third of the variability in fire size, regardless of weather and other effects. The lowest R^2 was measured for open forest and low, dry open woodland, reflecting the influence of a small number of remaining outliers rather than a weak trend (Fig. 3).

Driving mechanisms

These findings are consistent with broad mechanistic expectations that have been proposed for angiosperm forests (Bond & Midgley 2012), and challenge the assumption that fuel accumulation will override other processes to produce negative feedbacks. Positive feedbacks have been shown to correlate with the development of a gap between canopy and understorey plants, and to both understorey thinning and succession to less flammable species that occurs with canopy growth (Kitzberger *et al.* 2012, 2016; Zylstra 2016). In addition to observed correlation, however, these mechanisms have been shown to be the primary endogenous determinants of fire behaviour in these forests, explaining 80% of the variability in flame heights observed within the four forest (non-woodland) communities during a major bushfire in 2003. In contrast, fuel load was able to explain only 11% of that variability (Zylstra *et al.* 2016).

This analysis adds to the empirical body of evidence, as such post-fire vegetation dynamics are common in the forests of the Alps (Fig. 6a,b). Feedback strength is likely to derive from the degree to which these factors change between post-disturbance and mature forests, so that the strongest feedbacks are evident in forests that progress from dense, flammable regrowth close to the ground, to mature stands with large separations between dense tree crowns and more open, less flammable understorey growth.

Table 4. Mean annual likelihood L of fire at a point in a formation

Formation	L	Standard error
Tall, wet forest	0.026	0.013
Open forest	0.025	0.014
Dry, open forest	0.032	0.012
Subalpine forest-woodland	0.023	0.019
Low, dry open woodland	0.032	0.016

Limitations

An implication of this is that less severe fire impacts may produce weaker feedbacks, because the difference is smaller between mature and post-disturbance

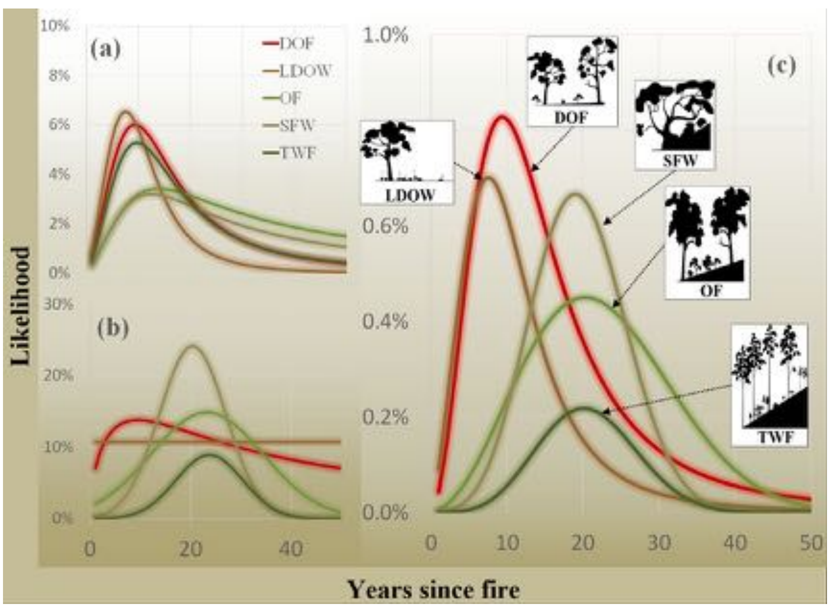

Fig. 5. Flammability trends for each formation, where the x-axis gives years since the last fire, and the y-axis gives likelihood for (a) fire burning a point (L_f), (b) crown fire occurring if that point is burning (L_{cb}) and (c) crown fire occurring at any point (L_c). Labels refer to dry, open forest (DOF), low, dry open woodland (LDOW), open forest (OF), subalpine forest and woodland (SFW), tall, wet forest (TWF).

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Fig. 6. Post-fire recovery of (a) dry, open forest with a canopy both resprouting and reseeding, and (b) tall, wet forest regenerating from seed alone. Both formations have produced abundant low fuel with little canopy protection. Severity effects vary, depending on the formation, so that in open forest, low severity fire can leave the canopy intact, but stimulate ground growth (c), whereas high severity can change both factors (d). In subalpine forest and woodland, both low (e) and high (f) severity tend to kill the canopy and promote ground growth, although to differing extents.

states. Weaker feedbacks may potentially occur where fires do not kill the crown, promote growth close to the ground, or do not produce a more continuous ladder of fuels between ground and canopy fuels, such as through the promotion of epicormic growth. It cannot yet be determined whether such fires can change the direction of feedback by creating a regeneration period that is less flammable than the mature period, but they may produce weaker positive feedbacks. The strength of this effect will depend on the formation in question, as structure and species' sensitivity to fire could have overriding effects (Fig. 6c–f).

The analyses in this study are also limited in that they describe historical trends only. If changing atmospheric and climatic influences act to vary species' dominance, structure or leaf traits such as thickness or chemistry, then the flammability of component species will also vary, with unforeseen effects on stand flammability. For example, (Prior & Bowman 2014) found that large eucalypt species are likely to grow more slowly in a warmer environment, and as the onset of the mature phase is related to the formation of a sufficient gap between canopy and lower plants, this may act to extend the flammable period of regeneration. Biophysical modelling is required to assess these effects and determine absolute flammability measures.

Implications

The findings of this study have significant implications for the Australian Alps, as they demonstrate that positive flammability feedbacks are not only present in the alpine area, but are pronounced and prevalent across the entire forested bioregion. Kitzberger *et al.* (2012) showed through spatial modelling that, where such feedbacks exist, increases in ignitions 'may, in synergism with increased climate variability, induce abrupt shifts and large-scale forest degradation.'

Such concern has already been raised with regard to the ash-type forests in the tall, wet forest formation (Bowman *et al.* 2014a; Burns *et al.* 2015), which are

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experiencing significant decline due to frequent fire, and to logging in areas adjacent to park. Two forms of transformation in these forests fit the definition of ecosystem collapse (Keith *et al.* 2013). Firstly, where the canopy is heavily dominated by the obligate-seeding ash, successive crown deaths within the primary juvenile period result in a distinct tipping point, and consequent transition to shrubland. This transition has been observed in several locations since 2003 (Bowman *et al.* 2014a; Wright & Robertson 2014). Secondly, where a resprouting sub-dominant canopy tree such as *E. dalrympleana* is present in sufficient numbers, frequent fire leads to a shift in formation from tall, wet forest to open forest. This second form of collapse is potentially well advanced already, although differences in survey technique introduce uncertainty in that determination. Early surveys found that *E. delegatensis* almost entirely dominated ash forest canopies in the NSW Alps (Byles 1932), yet data from recent surveys (Wright & Robertson 2014) showed dominance in only 68% of sites in mapped ash forest, with other species typical to the open forest formation dominating the remaining fraction.

Transition from tall, wet forest to open forest introduces changes to landscape flammability in the Alps that may be considered permanent on a scale of human management, as mature open forest is more flammable than mature tall, wet forest (Table 2). In terms of L_f and L_c (Fig. 5a,c) averaged for these mature periods, this represents a doubling in the likelihood of fire at any point, and four times the likelihood that any point will experience crown fire in a given year. This has potential to increase fire contagion and transform the flammability of the entire bioregion.

Concluding remarks

This study constitutes the first examination of fire-flammability feedbacks for these forests, and thereby provides a basis by which fire risk assessment and mitigation can transition from assumed feedbacks to an evidential basis. Across the Australian Alps, recently burnt forests have been on average more flammable than mature forests, consistent with historic observation and the mechanistic understanding arising from plant growth and species' change. Increases in fire frequency are therefore likely to create a more flammable landscape, with implications for both natural and built assets. Drivers of post-fire succession such as fire severity or expected climatic and atmospheric changes may vary the pattern or strength of these feedbacks, but this has yet to be shown through either empirical measurement or mechanistic modelling.

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SUPPORTING INFORMATION

Additional Supporting Information may be found in the online version of this article at the publisher's web-site:

Appendix S1. Data used to calculate the flammability ratio.

Appendix S2. Data used to calculate crown fire likelihood.

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