Effects of Sydney Coastal Dry Sclerophyll Forest Litter on Fuels and Fire Behaviour in Hornsby Shire

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This is to certify that to the best of my knowledge, the content of this thesis is my own work. This thesis has not been submitted for any degree or other purposes.

I certify that the intellectual content of this thesis is the product of my own work and that all the assistance received in preparing this thesis and sources have been acknowledged.

Signature

Name

Angela G. Gormley

O! for a muse of fire, that

would ascend the

brightest heaven of

invention

Prologue

The Life of King Henry V

William Shakespeare

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Abstract

Globally bushfires are an ecological phenomenon that can cause deaths and widespread destruction of assets such as homes, utilities and essential infrastructure. Bushfires usually start in forest litter on a forest floor. The research described in this thesis used empirical data to characterise the physical and chemical attributes of litter, a component of forest and woodland fuels that is particularly important for propagation of fire. Differences in the amounts, arrangement and flammability of components of litter were determined for Sydney Coastal Dry Sclerophyll Forest, a common vegetation type in the Sydney Basin. Surface litter was investigated at study sites at Rofe Park, Hornsby Heights and Halls Creek, Arcadia, New South Wales, Australia. Data describing fuel load, structure and condition of surface litter were gathered using semi-quantitative (fuel hazard score, percent cover score, pin transect) and quantitative methods (surface litter depth, bulk density and soil moisture). These methods indicated both the Halls Creek and Rofe Park sites had ladder fuels with an extreme risk of fire. Chemical analyses of litter fractions included total carbon and nitrogen. The carbon content of litter was similar between sites but there was a significant difference in nitrogen because of the presence of the nitrogen-fixing species, *Allocasuarina littoralis*, at Rofe Park. This site was also wetter compared to Halls Creek. However, these physical and chemical parameters provide little information about flammability of surface litter from these sites. Surface litter was sorted into fractions (e.g. whole leaves and twigs, partially and fully decomposed organic material) and used to determine which component or mixture of components were the most flammable. The Simplex Centroid Design (SCD) method was used to determine optimum mixtures of fuel fractions and a General Blending Model was used to determine the best statistical model fit for flammability metrics (ignitability, combustability, consumability and sustainability). Flammability measures included time to ignition, burn to completion, vertical fuel height, rate of spread, volume consumed, duration of vertical flame and residual mass fraction. Fuel from both sites had rapid time to ignition although, 60% of Halls Creek samples failed to burn compared to 35% of the Rofe Park mixtures. Surface fuels from Rofe Park proved to be more flammable than those from Halls Creek because visual flame heights were twice as high and rate of spreads were twice as rapid. *Allocasuarina littoralis* provided a non-additive effect by driving flammability of Rofe Park litter mixtures. Halls Creek samples had negative and positive non-additive effects for three flammability metrics; bulk density, residual mass fraction and rate of spread. Rofe Park samples had strong positive non-additive effects of six flammability metrics; burn to completion, residual mass fraction, rate of spread, volume consumed, vertical fuel height and duration of vertical flame. *Allocasuarina littoralis* was the most flammable component overall and twigs were the most flammable component in the litter from Halls Creek. The data were optimised to find the maximum or minimum fit for the SCD. These optimisations demonstrated the ideal litter mixtures from both sites that produce the maximum and minimum flammability for all flammability metrics. Knowledge about fuel flammability of litter from sites with the same fire-prone vegetation type will help inform management decisions about prioritising prescribed burning to mitigate the risk of fire. Future studies could investigate the use of the experimental methodology presented in other forest types.

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Chapter One: Overview of fires globally and in Australia

Globally, bushfires are an ecological force often with widespread impact. Extensive bushfires have been reported in Australia (Valent 1984; Ganewatta 2008; Leonard *et al.* 2009), North America (Cohen 2000) and Europe (Lampin-Maillet *et al.* 2010). A major bushfire can cost hundreds of millions of dollars because of fire suppression, insurance costs, deaths, damage to homes and urban infrastructure including electricity supplies, water, roads and railways (Gentle *et al.* 2001; Gillen 2005; Handmer *et al*. 2018). However, treatments to mitigate the risk of bushfires are expensive so the land manager must weigh up the level of risk to assets with costs for preventive measures. For example, the average annual fire suppression costs in the Hornsby/Ku-ring-gai District in New South Wales (NSW), Australia, are estimated at $5.95 million per year (Australasian Fire and Emergency Service Authorities Council 2018).

Official enquiries into devastating bushfires such as the 'Black Saturday Fires' in Victoria in 2009 have prompted investigations to prevent future property damage and loss of life. The recommendation to increase the areas of prescribed burns is a common response (Esplin *et al.* 2003). For example, Recommendation 56 of the *2009 Victorian Bushfires Royal Commission* was to increase the amount of prescribed burning to at least 5% of public land annually (Victorian Bushfires Royal Commission 2010). Fuel management policies in Victoria now enable larger areas to be treated with prescribed burning to reduce fuel loads and to potentially save lives and strategic assets. The risk-based approach for prescribed burning sits within broader risk management processes implemented nationally (AS/NZS ISO 31000-2009; Standards Australia 2009) (Inspector General for Emergencies

Management 2015). However, bushfire mitigation treatments cannot be used on all fire-prone vegetation because the areas are too large and the costs are prohibitive and the treatment is not suitable for the vegetation type (e.g. Ash-type forests which are often too wet to prescribe burn) or landscape features (e.g. rugged, hilly terrain). A better understanding of flammability and fire behaviour within fire-prone forest classes will help land managers prioritise fire mitigation treatments. Land managers need to have integrated fuel treatment planning and optimisation models that are easy to learn and use while providing practical applications in the forests they manage (Chung 2015).

1.1 Characterisation of vegetation in the Sydney Basin

Vegetation in the Sydney Basin is determined by soil type, soil moisture, rainfall amount and seasonal distribution, wind exposure or shelter and fire frequency. Vegetation structure, including ground cover, understorey and overstorey species, tree height and closure or openness of the canopy, can influence fire behaviour (Gould *et al*. 2011). Typical vegetation types in the Sydney Basin include heath, scrub, woodland, open forest, tall open forest and small pockets of rainforest (Fairley and Moore 2010). Heath communities are located on coastal sands and shallow sandy soils (Keith 2004) and are characterised by shrubs lower than 2 m tall. Open heaths have foliage cover of 30–70% whereas closed heaths have coverage of 70–100% (Fairley and Moore 2010). Scrub is typically found on sandy soils on ridges and slopes along swamp sedgelands such as on the Blue Mountains plateau, with density and height of these shrubs being dependent on fire frequency, drainage and soil type. This vegetation type is 2–8 m tall without a tree canopy and with densely growing sclerophyllous plants (Fairley and Moore 2010). Woodlands are frequently found in western ranges of valleys, on top of plateaux and in slopes and in areas with low rainfall including the Cumberland Plain (Keith 2004). There can be a species-rich understorey but the structure and composition can vary greatly. Coastal low open woodlands are dominated by eucalypts with open canopies of 10–30% cover (Fairley and Moore 2010). Open forest dominates most areas of the Sydney Basin (Keith 2004) with trees up to 30 m having a canopy cover of 30–70% and an understorey of ground cover and shrubs. Tall open forests are associated with rich fertile basalt soils and sheltered slopes and gullies (Fairley and Moore 2010). The trees are 30 m or greater with a canopy cover of 30– 70% and an understorey consisting of vines, shrubs, ferns and herbs. Rainforests have closed canopies of 70–100% (Keith 2004) and there are generally only a few shrubs and ferns in the understorey (Fairley and Moore 2010).

Some of these forest types are not considered to be fire-prone but will burn during a bushfire. For example, several studies describe mechanisms that rainforest species have to cope with fire including, fire-resistance (Baker *et al.* 2012; Clarke *et al.* 2014), post-fire resprouting (Melick and Ashton 1991; Benson and McDougall 1997; Williams 2000; Prior *et al.* 2007; Clarke *et al.* 2014) and seed germination in gaps caused by fire (Hill and Read 1984; Williams 2000; Jennings and Neyland 2011).

1.2 Flammability of biomass

In forests and woodlands litter is the primary fuel for surface fires and can influence fire behaviour due to its chemistry, ubiquity and mass (Gill and Zylstra 2005; Varner *et al.*

2015). Surface fuels include live and dead grasses, senesced leaves, woody debris, forbs and shrubs (Keane 2015). Fuel arrangement, structure, species composition, moisture content and state of decay collectively influence ignition, fire intensity, rate of spread and fuel consumption (Varner *et al.* 2015), all of which are affected by or contribute to fuel 'flammability' (Anderson 1970; Martin *et al.* 1994; Gill *et al.* 2005). Bushland flammability describes the capacity of forest litter fuels to ignite and combust (Varner *et al*. 2015).

To determine the flammability of biomass fuel, including that representing forest types in the Sydney Basin, fuel properties such as calorific value, ash content, and volatile content (Nordin 1994; Dickinson and Kirkpatrick 1985). Calorific content of fuel is determined by how much heat is produced from a unit mass of fuel (MJ kg^{-1}) (Kumar *et al.* 2010). For example, biomass with a high calorific content, low ash content, high fixed carbon content and high density has high energy output per unit volume and tends to burn slowly (Singh and Khanduja 1984; Goel and Behl 1996; Kumar *et al.* 2010). The carbon content of different tree species can vary considerably (Table 1.1; Snowdon *et al.* 2005). The flammability of fuel depends on the main chemical components including cellulose, hemicellulose, lignin and extractives and ash- forming minerals (Shafizadeh 1982), which can also vary considerably according to species (Table 1.2).

Amounts of carbon and nitrogen in forest litter are often reported on an ash-free basis after dry combustion of the sample (Snowdon *et al.* 2005). Forest ecosystems in the Sydney Basin, as in other parts of Australia, are limited by nitrogen where ecosystem productivity is influenced by nitrogen turnover rates, nitrification and mineralisation in forest soils (Adams and Attiwill 1986; Vitousek *et al.* 2002; Galloway *et al.* 2004). *Eucalyptus* forests, woodlands and plantations have lower nitrogen values than vegetation dominated by *Acacia*, which is a nitrogen-fixing species (Tables 1.3 and 1.4, respectively).

Carbon to nitrogen ratios (C:N) have long been recognised as useful indicators of the source of organic matter, the state of decomposition and possible contribution to soil fertility (e.g. Alexander 1977; Swift *et al.* 1979; Paul 2007). A ratio higher than 25, on a mass basis, demonstrates that decomposition is slower than the rate of accumulation of litter (Bui and Henderson 2013). A lower C:N value means the forest is more productive (Snowdon *et al.* 2005). Leaching can lead to more effective litter decomposition (Snowdon *et al.* 2005), causing high C:N values. In general, C:N ratios are lower for nitrogen-fixing species than for non-nitrogen fixing species and the nitrogen range of litter from *Eucalyptus* forests is between 64 and 112% (Snowdon *et al.* 2005) (Table 1.1).

Table 1.1 Carbon (C):nitrogen (N) ratio in surface soil layers for various forest ecosystems. Calculated values are derived from the database summaries derived from individual data (Adapted from Snowdon *et al.* 2005). Min – minimum; Max – maximum.

Ecosystem	Published soil C:N ratio				Calculated soil C:N ratio			
	n	Mean	Min	Max	n	Mean	Min	Max
Native forest								
Sclerophyll	16	32.6	20.7	54.4	16	31.7	21.0	49.5
Dry Sclerophyll	22	32.4	18.4	59.8	91	32.6	3.0	59.6
Woodlands								
Woodland	0				2	33.6	24.8	42.4
Mixed eucalypt woodland	7	23.3	16.0	32.1	7	22.9	15.8	31.7
Open woodland	0				2	13.8	12.9	14.6
Leptospermum	1	22.3	22.3	22.3	1	22.3	22.3	22.3
Acacia	1	23.6	23.6	23.6	1	23.7	23.7	23.7

Table 1.2 Carbon (C) content of wood calculated from extractive analysis data (Source: Snowdon *et al.* 2005).

Component	*Eucalyptus* sp.	Blue Gum	River Red Gum	Black Wattle
Ash	0.24	0.3	1.7	0.9
Cellulose	47.9	51.3	45.0	42.9
Hemi-cellulose	32.2	22.3	19.2	33.6
Lignin	25.1	21.9	31.3	20.8
Other organics	1.9	1.3	2.8	1.8
Estimated [C]	49.7	49.3	50.7	48.8

Table 1.3 Nitrogen concentrations (% oven dry weight) in litterfall from various forests and plantation ecosystems (Source: Snowdon *et al.* 2005). NSW – New South Wales; QLD – Queensland.

Ecosystem	Mean	Minimum	Maximum	C:N	n
Eucalypt, coastal NSW/QLD	0.59	0.30	0.94	100	7
Eucalypt woodland	0.67	0.47	0.89	78	4
Eucalypt plantation	0.81	0.40	1.32	67	13
Acacia plantation	1.81	1.17	2.24	30	7

Table 1.4 Nitrogen concentration (% oven dry weight) and C:N ratios in litter from various native forest and plantation ecosystems (Source: Snowdon *et al.* 2005). NSW – New South Wales; QLD – Queensland.

Ecosystem	Mean	Minimum	Maximum	Mean C:N	n
Eucalypt-southern QLD	0.74	-	-	73.00	10
Eucalypt-coastal NSW/QLD	0.63	0.36	0.78	86.00	5
Eucalypt-inland NSW	0.42	0.30	0.63	1.32	3
Eucalypt plantation	0.71	0.27	0.95	84.00	6
Acacia dealbata plantation	2.17			23.00	1

Table 1.5 Calculated carbon (C):nitrogen (N) ratios for surface litter layers in various ecosystems (Adapted from: Snowdon *et al.* 2005).

Ecosystem	n	Mean	Minimum	Maximum
Dry sclerophyll forest	11	88.8	58.4	148.0
Mixed eucalypt woodland	7	61.6	36.8	94.0

1.3 Forest litter

1.3.1 What is forest litter?

Trees constantly shed leaves, twigs, branches, hard woody fruits, bark and roots. As soon as the plant material falls off a tree it is called litter (Berg and McClaugherty 2013). Leaves are usually the dominant component and can be 41–98% of total forest litter by volume (Benson and Pearson 1993; Abelho and Graça 1996; Abelho and Graça 1998; Grigg and Mulligan 1999; Oelbermann and Gordon 2000). However, the composition of forest litter varies with vegetation type and location, so non-leaf components of forest litter can be as high as 78–80% by volume in some Australian *Eucalyptus* forests (Hart 1995). Bark can make a significant contribution to forest litter (Gould *et al*. 2011). Ironbarks (e.g. *Eucalyptus tricarpa* and *E. paniculata*) have bark that is smooth on the whole tree trunk (Gould *et al*. 2011) and can be shed on an annual basis. Smooth/candle barks (e.g. *E. rubida*, *E. rossi, E. globulus* and *E. regnans*) shed long ribbons of bark and trunks can be smooth down to their base (Gould *et al*. 2011). Stringybarks (e.g. *E. marginata*, *E. obliqua* and *E. baxteri*) shed long loose flakes of bark from the whole tree trunk down to the ground but in far smaller quantities than smooth-barked species.

Litter can be described as dead plant material in various stages of decomposition located above the mineral soil with a minimum cross-sectional dimension of less than 25 mm (Volkova and Weston 2013; Bennett *et al*. 2014; Volkova and Weston 2015; Jenkins *et al.* 2016). Chemically, litter mostly consists of lignin, cellulose and hemicellulose. Fine fuel in the surface and near-surface layers, a category of litter often used by land managers in Australia, is defined as material less than 6 mm in diameter (Hines *et al.* 2010; Gould *et al*. 2011).

1.3.2 What does litter do?

Litter plays a vital role in soil ecosystems. Plant litter contains essential nutrients that enable other organisms to grow (Berg and McClaugherty 2013) and includes nitrogen, magnesium, manganese, potassium, sulphur, phosphorus, calcium and iron. Until forest litter is degraded these nutrients are 'locked up' in organic biomass. Forest litter can modify water and energy exchange between forest soil and the atmosphere (Ogee and Brunet 2002). Forest litter cover can prevent water from penetrating into soil, can deflect most radiation and lowers soil evaporation (Kelliher *et al.* 1986; Putuhena and Cordery 1996; Schaap and Bouten 1997; Ogee *et al.* 2001). The changes effected by litter can play a significant role in carbon cycling since soil moisture and temperature determine soil respiration (Schlentner and Van Cleve 1985; Cropper and Gholtz 1991; Hanson *et al.* 1993; Keith *et al.* 1997; Davidson *et al.* 1998; Boone *et al.* 1998; Rayment and Jarvis 2000). Fresh and decomposing forest litter produce heat, water and carbon dioxide. The rate of evaporation of water from litter is regulated by temperature, humidity and the rate of decomposition of fresh litter, which, in turn, affects the production of carbon dioxide (Keith *et al.* 1997; Schaap and Bouten 1997; Gunadi *et al.* 1998; Pulleman and Tietema

1999).

Microorganisms are the most important decomposers of litter (Berg and McClaugherty 2013). Fungi and bacteria release carbon dioxide and dissolved organic compounds from litter and physically break down plant material into decomposed litter. Microorganisms are the catalysts for reduction-oxidation reactions which drive the biogeochemistry of nitrogen (Falkowski 1997; Canfield *et al.* 2010). Invertebrates such as earthworms are involved in turnover or comminution of litter. Vertebrates play an important role in spreading fungal spores of macrofungi in forests and woodlands. More than fifty species of fungi have been found in scats of mycophagous mammals (Claridge and May 1994; Johnson 1995).

Fungi are an important component of forest organisms as they influence fuel accumulation by decomposing forest litter, coarse woody debris and logs. The most important of these is saprotrophic fungi that grow in soil, litter or wood (Rayner and Boddy 1988; Bridge and Spooner 2001). Saprotrophic, or decomposer fungi, have the ability to break down the most complex compounds including cellulose and lignin (McMullan-Fisher *et al.* 2011). It has long been accepted that competition is particularly intense between fungi for saprophytic colonisation of a fresh forest litter substrate (Garrett 1951) and dead and decaying wood (Rayner and Todd 1980; Boddy 2000). There have been few studies of how fires affect soil fungi but it is well known that fire tends to alter the community structure of fungi and the effect is greater with frequent burning (Cairney and Bastias 2007).

1.4 Coarse woody debris

Coarse woody debris (CWD) is any dead plant material that has a cross-sectional dimension greater than or equal to 6 mm (Hines *et al.* 2010; Gould *et al.* 2011). The distinction between fine fuel and CWD in forest litter is useful since fire behaviour and carbon emissions are dependent on the fuel source. Fire intensity and carbon emissions increase as CWD is consumed. Fine fuels can combust com

pletely within minutes releasing mostly carbon dioxide but coarse fuels can smoulder for days with higher emissions of incompletely oxidised products such as methane, carbon monoxide, volatile organic compounds, nitrogen oxides (Volkova *et al.* 2014). For example, the methane emission ratio from coarse fuels can be up to eight times greater than from flaming combustion of litter (Volkova *et al.* 2014). A low intensity prescribed fire can be responsible for emitting up to 3.74 Mg C ha^{-1} of carbon dioxide, 0.40 Mg C ha^{-1} of carbon monoxide and 0.024 Mg C ha^{-1} of methane, while about 2.3 Mg C ha^{-1} may be released from other carbon pools (understorey, CWD, ground cover and deadwood) (Volkova and Weston 2015).

Coarse woody debris is an important regulator of carbon emissions since many forests have a significant proportion of CWD and there is great variability in fire behaviour affecting the amount of carbon released. Charred CWD is a pyrogenic carbon source that is produced from forest fires and deposited in soils. Typical amounts range from 1–2% of biomass and 1–10% of available biomass (Tinker and Knight, 2000; Lynch *et al*. 2004; Forbes *et al*. 2006; Preston and Schmidt, 2006; DeLuca and Aplet, 2008). Variations in values can be caused by the amount of CWD in the forest and how much is consumed during a fire event

– between 9 and 89% can be consumed during planned fires (Hollis *et al*. 2010), and this increases to 100% for bushfires (Hollis *et al*. 2011). Residues are deposited after fire in the form of charcoal, ash, char, black carbon and pyrogenic carbon (Jenkins *et al.* 2016).

Estimates of carbon emissions have been based on fuel loads by measuring fine fuels. In Australia, fuel load measurements have been an important aspect of fire behaviour models. However, recently authors have questioned their effectiveness in determining carbon emissions from fires. Possell *et al.* (2015) and Volkova *et al.* (2015) have pointed out that CWD and understorey fuels are also burnt during forest fires so their omission from these models will cause inaccuracies.

1.5 Fire effect on fuel loads

Prescribed burns reduce fuel loads by removing forest litter (surface fuel). Internationally, surface fuel load is a common descriptive characteristic that is based on the dry weight biomass of fuel per unit area in kg m^{-2} (Pyne 1996; Keane 2013). However, prescribed burns only tend to be effective for a short time because of the rate of accumulation of forest litter on the forest floor (Raison *et al.* 1983; Morrison *et al*. 1996). For example, in the study of Morrison *et al.* (1996) in the Ku-ring-gai Chase National Park, NSW, the steady state fuel load was estimated to be approximately 32 and 42 t ha^{-1} for shrubland and woodland communities, respectively. Planned, fuel-reduction burns are conducted approximately every 5 years in the Sydney region but within this time fuel loads of 13–19 t ha^{-1} can accumulate (Morrison *et al.* 1996). Similarly, in *Eucalyptus piluaris* forests in northern NSW, fuel loads were the same as prior to prescribed burning within 4 years (20 t ha^{-1}; Birk and Bridges 1989). Unburnt *E. obliqua* forests in Victoria have maximum litter loads of 25.9 t ha^{-1} (Hamilton *et al.* 1991). Likewise, the litter fuel load in dry eucalypt forest in Tasmania was about 4 t ha^{-1} after 4.5 years after fire (Prior *et al.* 2016). All of these landscapes seem to be at risk since fuel loads of 0.8–1.0 t ha^{-1} are necessary to support a fire (Burrows *et al.* 1990).

Fuel load influences fire behaviour through its link to fire intensity (Byram, 1959). Fire intensity is defined as kW per m of fire front (Luke and McArthur 1978; Cheney and Sullivan 1997). During the Christmas 2001 Sydney bushfires there were fire intensities of more than 50,000 kW m^{-1} in areas with fuel loads exceeding 40 t ha^{-1} that had not been burnt since 1965 (Chafer *et al*. 2001; 2004). However, the bushfire still destroyed vegetation with fire intensities greater than 7,000 W m^{-1} in areas that had fuel reduction treatment the previous season and a fuel load less than 5 t ha^{-1} (Chafer *et al.* 2004). The Mount Hall and Nattai bushfires in NSW during December 2001 to January 2002 had the greatest fire intensity on ridges where significantly higher fuel loads are found on plateau tops and was lower in valleys (Chafer *et al.* 2004; Bradstock *et al.* 2010). The probability of an understorey fire is reduced within 1–5 years of a prescribed burn but is substantially increased after 5–10 years (Bradstock *et al*. 2010). Fires that can be suppressed are more likely to occur where fuel loads have been reduced via prescribed burning within 1–5 years and crown fires are prevented (Bradstock *et al.* 2010).

1.5.1 Consequences of fire for carbon and nitrogen cycling

Fires in forests affect carbon and nitrogen cycling. Land managers need quantitative data to determine how to do prescribed burns while decreasing carbon emissions, carbon stock losses and greenhouse gas emissions from biomass fires (van der Werf *et al.* 2006; Bowman *et al.* 2009; Volkova and Weston 2013; Bennett *et al.* 2014; Volkova and Weston 2015; Jenkins *et al.* 2016). However, when emissions from prescribed burns are greater than from bushfires then fuel emissions will not be reduced (Bradstock and Williams 2009; Mitchell *et al.* 2009; Campbell *et al.* 2011). Temperate eucalypt forests have been estimated from the Net Ecosystem Exchange to sequester carbon at a rate of 2–6 t ha^{-1} $year^{-1}$ in the period between fires and lose carbon from fires at a rate of approximately 1.0–1.7 ha^{-1} $year^{-1}$ so the Net Biome Productivity (NBP) is about 1–5 t ha^{-1} $year^{-1}$ (van Gorsel *et al.* 2008; Keith *et al.* 2009; Bradstock *et al.* 2012a). Thus, prescribed burns in Australian *Eucalyptus*-dominated temperate forests are unlikely to produce a net carbon reduction in carbon emissions because the prescribed burning leverage is too low since these fires will further reduce the NBP and sequestration capacity of these forests (Boer *et al.* 2009; Bradstock *et al.* 2012 a, b).

Forests play an important role in carbon and nitrogen emissions and sequestration of carbon in soils (Stockmann *et al.* 2013). Fires emit trace gases into the atmosphere (Paton-Walsh *et al.* 2014) – emissions for Australian temperate forest fire are shown in Table 1.6. Since the total quantity of emissions from biomass fire varies greatly each year, this affects the changing composition of the troposphere affecting carbon and nitrogen cycling (Paton-Walsh *et al.* 2014). The major carbon losses are caused by bushfires, timber harvesting and prescribed fire (Volkova *et al*. 2015). Fire causes several processes that emit carbon: combustion, transfers between pools from live to dead biomass, and regeneration of vegetation after fire (Bennett *et al*. 2014; Volkova and Weston 2015). Severity, intensity and interval between fires affects carbon storage in soils (Volkova and Weston 2013; Bennett *et al*. 2014; Volkova and Weston 2015). For Victorian forests in Australia, it has been estimated that the canopy contributes 50–70% of aboveground biomass and the understorey from 8–33% and there is a significant and positive correlation between amount of aboveground biomass and the amount that is burnt by fire (Volkova *et al.* 2015). It stands to reason that more carbon is lost when there is more biomass. Erosional losses and plant inputs to soil from regeneration and growth are also affected by post-fire interactions in climate and topography (Keeley 2009; Bennett *et al.* 2014).

Table 1.6 Emission factors for Australian Temperate forest fires expressed as mean (± standard deviation) grams of gas emitted per kilogram of dry fuel burned (Source: Paton-Walsh *et al.* 2014).

Gas	Emission values (g kg^{-1})
Carbon dioxide	1620 ± 160
Carbon monoxide	120 ± 20
Methane	3.6 ± 1.1
Ethylene	1.3 ± 0.3
Formaldehyde	1.7 ± 0.4
Methanol	2.4 ± 1.2
Acetic acid	3.8 ± 1.3
Formic acid	0.4 ± 0.2
Ammonia	1.6 ± 0.6
Nitrous oxide	0.15 ± 0.09
Ethane	0.5 ± 0.2

1.6 Prescribed burning objectives

1.6.1 Ecological aims

Prescribed burning is an important ecological tool for land managers when it is used to manage ecosystems to promote biodiversity (Bradstock and Kenny 2003; Bowman *et al.* 2016). This type of management is necessary to enable plants to germinate, survive and to promote plant diversity in forests (Tremont and McIntyre 1994; Morgan 1998; Lunt and Morgan 2002). Native plant communities can be infested with invasive flammable weeds, such as *Lantana camara*, in the understorey of forests in south-eastern Australia (Duggin and Gentle 1998; Jeffs 2004). Land managers take into account the needs of plant and animal communities to plan and manage the intensity and frequency of fires. For example, certain species respond to germination cues related to fire and produce many seedlings post-fire while other species may require shade and deep litter for their seedlings to survive (Poulsen and Platt 1989; Auld and O'Connell 1991; Hoffmann 2000; Bell *et al.* 1993; Tang *et al.* 2003; Clarke *et al.* 2010; Williams *et al.* 2012; Lusk *et al.* 2013). There are competing arguments for and against prescribed burning caused by conflicting fire management, economic, environmental and social values intertwined with uncertainties and complexities in fire ecology (Gillen 2005; Bowman *et al.* 2016). The best way to sustainably manage fire-prone forests, woodlands and shrublands is very much open to debate (e.g. Bradstock and Kenny 2003; Donovan and Brown 2007; Reinhardt *et al.* 2008; New *et al.* 2010; Bowman *et al.* 2016; Jenkins *et al.* 2016).

1.6.2 Risk management of prescribed burning

Land managers in Australia implement prescribed burns (also often referred to as hazard or fuel reduction treatments or burns) to manage the risk of bushfires by reducing fuel loads (Ellis *et al.* 2004). Despite considerable research and management efforts, there is no consensus about the extent that prescribed burning mitigates fire risk by reducing fuel loads (Fernandes and Botelho 2003; Ellis *et al.* 2004). Reducing fuel loads is supposed to lower fire intensity to increase the likelihood of successful fire suppression (Raison *et al*. 1983; Morrison *et al*. 1996). Recent research has demonstrated that bushfires are most intense on ridges so prescribed burning of these areas could have strategic value instead of burning landscapes (Bradstock *et al.* 2010).

Fire management plans in NSW usually assess bushfire risk using a combination of local knowledge and GIS modeling following the methods of Dovey (1994) or Rose and Chafer (1997) or a combination of both (e.g. Bean and Jones 2001; Chafer *et al.* 2001; Bean 2002; Bushfire and Environmental Services 2002). Landscapes can be classified as low, medium or high risk of bushfires and model inputs are slope, aspect, fuel loads, vegetation patterns and fire history (Rose *et al*. 1999). These models provide reliable and rapid assessments of the risks that are essential for bushfire management decisions on the urban-bush interface.

In the Sydney Basin, locations at risk of bushfires are often densely populated areas located near fire-prone vegetation. A classic example is in the Shire of Hornsby, located in the Sydney Basin north of Sydney, where this research is located. The area of Hornsby Shire is 462 km^2 with 70% under environmental protection or National Park (Hornsby Shire Council 2016). There is a population of 142,667 with 39,339 families and 51,240 private dwellings (Australian Bureau of Statistics 2016). Over one third (31%) of this population lives within 130 m of the urban-bush interface (Chen 2005; Hornsby Shire Council 2016). The Rural Fire Service (RFS) provides guidelines to Councils for mapping Bush Fire Prone Land (BFPL) (Rural Fire Service 2015). In addition, a broad-scale classification and map of the native vegetation in NSW (Keith 2004) is used. This statewide description of native vegetation has three levels of a hierarchy: formations, classes, and communities. The vegetation classes of Keith (2004) are used to classify vegetation into BFPL categories. Most areas in Hornsby Shire are classified as BFPL Category 1 which is the highest risk (Hornsby Shire Council 2018).

Hornsby Shire has been identified as an area at risk of bushfires. In Section 52 of the NSW *Rural Fires Act 1997* the Hornsby/Ku-ring-gai Bushfire Management Committee is required to provide the Bushfire Coordinating Committee with a Bush Fire Risk Management Plan (BFRMP) (Hornsby/Ku-ring-gai Bush Fire Risk Management Plan 2016-2021). This BFRMP identifies community assets that are at risk within the categories of human settlement, economic, environmental and cultural. This includes residential and rural properties on the urban-bush interface. For these asset types, the likelihood of a bushfire has been assessed on the basis of fire history, ignition cause and patterns, known fire paths, access containment, potential fire run based on the size of the vegetated area and vulnerability of human populations (Hornsby-Ku-ring-gai BFRMP 2016–2021). The level of risk to these assets is based on likelihood and consequence ratings. The proportion of at-risk assets in Hornsby Local Government Area was calculated from Hornsby/Ku-ring-gai BFRMP 2016–2021, Appendix 2 Asset Register. These proportions are: 'Extreme' 41%; 'Very high' 15%; 'High' 21%; 'Medium' 12% and 'Low' 11%.

Treatments to mitigate the risk of bushfires are expensive so the land manager must weigh up the level of risk to assets with the costs. For example, in the Hornsby/Ku-ring-gai District, the prescribed burning costs for the NSW RFS are approximately \$912 ha^{-1} for interface burning and for the NSW National Parks and Wildlife Service (NSW NPWS) costs are estimated at \$1,016 ha^{-1} (Australasian Fire and Emergency Service Authorities Council 2018). In Hornsby Shire, only assets in the 'Extreme' or 'Very high' risk categories have the application of treatment strategies as defined within the BFRMP.

The level of acceptable risk was 'High' because of a limit on time and resources (Hornsby-Ku-ring-gai BFRMP 2016–2021).

1.6.3 Importance of flammability in assessing risk

Fire managers refer to litter as 'fuel' and the amount of litter built up on the forest floor is quantified as the 'fuel load' which is an important flammability metric for managing fire risk to properties. Internationally, surface fuel load is a common descriptive characteristic that is based on the dry weight biomass of fuel per unit area in kg m^{-2} (Pyne 1996; Keane 2013). Land managers use information about fuel load, heat of combustion and rate of spread of fire to calculate the fire intensity using Byram's equation. Byram's (1959) definition of fire intensity is either the rate of energy or heat release per unit time per unit length of fire front:

$$I = H\,w\,r$$

Where I is fire intensity in kW m^{-1}, H is the heat of combustion in kJ kg^{-1}, w is the weight of fuel consumed per unit area in kg m^{-2}, and r is the rate of spread in m s^{-1}. Low intensity fires are less than 0.055 MW m^{-1} and high intensity fires are above 4 MW m^{-1} (McArthur and Cheney 1966; Sando 1978; Alexander 1982). Fire intensity provides the best description of the effects of fire on vegetation and soil, with the proviso of stationary spreads (McArthur and Cheney 1966; Burrows 2015). Duration of combustion is an essential fuel attribute. Even though advances in technology such as remote sensing and satellites have assisted in measuring the effect of fire at the landscape-scale, contemporary fire literature fails to provide equations for fire behaviour that are useful for quantifying landscape-scale responses of vegetation (McArthur and Cheney 1966;

Burrows 2015).

This lack of standards or test procedures causes contradictory results and there are restrictions in relating laboratory results with field conditions (White and Zipperer 2010). Standards have been established for the regulation of flammability of commercial products such as building materials (Apte 2006), however, there are no standards for testing whole plants or plant parts (White and Zipperer 2010).

Table 1.7 Selected standard test methods for flammability and combustion properties of materials (Source: White and Zipperer 2010).

Common name	Organisation	Designation	Title of test standard
Oxygen bomb	ASTM	D 5865	Test method for gross calorific value of coal and coke
	ISO	1716	Reaction to fire tests for building products – determination of the heat of combustion
Ignition	ISO	5657	Reaction to fire tests – ignitability of building products using a radiant heat source
Mass loss calorimeter	ASTM	E 2102	Test method for measurement of mass loss and ignitability for screening purposes using a conical radiant heater
Oxygen index	ASTM	D 2863	Test method of measuring the minimum oxygen concentration to support candle-like combustion of plastics (oxygen index)
Cone calorimeter	ASTM	E 1354	Test method for heat and visible smoke release rates for materials and products using an oxygen consumption calorimeter
	ISO	5660-1	Reaction to fire tests – heat release, smoke development and mass loss rate – part 1. Heat release rate (cone calorimeter method).
Full-scale calorimeter	ASTM	E 2067	Practice of full-scale oxygen consumption calorimetry fire tests.
Furniture	ASTM	E 1822	Test method for fire testing of stacked chairs
Room/corner	ASTM	E 2257	Test method for room fire test of wall and ceiling materials and assemblies
	ISO	9705	Fire tests – full scale room test for surface products
Micro-calorimeter	ASTM	D 7309	Test method for determining flammability characteristics of plastics and other solid materials using microscale combustion calorimetry

Information about fuel loads and flammability can be used to guide land managers in mitigation of risk from bushfires. In addition, some fire behaviour characteristics can be determined from the physical and chemical attributes of a plant (White and Zipperer 2010). However, there is a lack of research evaluating litter flammability by using both physical and biochemical traits across different plant species even though both of these components individually and interactively have been considered to be important for decades (Rundel 1981; Varner *et al.* 2015). A developing research area relates functionally important traits of plants to variability in leaf flammability (Schwilk and Caprio 2011). For example, the rate of ignition of fuel influences plant flammability and fire behaviour (Pickett *et al.* 2009) and leaf chemistry drives flammability (Varner *et al.* 2015). In the study of Ormeno *et al.* (2009), terpene concentrations were found to be broadly related to flammability. In another study, de Magalhaes and Schwilk (2012) found consistent non-additive effects in flammability of forest litter mixtures when determined by the species with the highest flames and most rapid rates of spread. Spread rate was closely related to leaf size and litter density probably because litter packing behaves non-additively as an influence on fire behaviour. However, this did not fully explain why litter mixtures had faster spread rates than for litter composed of individual species and this seemed to be caused by the most flammable species in the mixture (de Magalhaes and Schwilk 2012). Other studies have suggested that mineral, silica and volatile content of litter could be chemical causes of flammability (Mutch and Philpot 1970; Philpot 1970; Alessio *et al.* 2008a, b; Ormeno *et al.* 2009).

Plants that are characteristic of different vegetation types in the Sydney Basin, such as Wet Sclerophyll Forest, Dry Sclerophyll Forest and Heath, differ in their leaf morphology and leaf flammability traits. Research from North America, Australia, Europe and South America has consistently found that plant species have different flammability metrics, even within species considered to be analogs of each other, with significant differences among species in similar environments (Fonda 2001; Scarff and Westoby 2006; Ormeno *et al.* 2009; de Maghalhães and Schwilk 2012; Thissell 2014; Weir *et al.* 2014). For example, Mediterranean shrubs showed population-level differences in how temperature was integrated and released during combustion which was putatively caused by variations in chemical composition (Pausas *et al.* 2012). Similarly, in an Australian study, broader leaves of exotic species ignited more rapidly compared to native species with smaller leaves. As a consequence, weedy plants infesting dry sclerophyll forests were likely to increase flammability of the whole ecosystem since larger leaves open the litter bed structure allowing better ventilation to make the fuel bed burn more rapidly (Scarff and Westoby 2006; Murray *et al.* 2013). Likewise, other studies have found that species such as *Quercus kelloggii*, *Pinus jeffreyii* and *Pinus ponderosa* with large or long leaves are more flammable than co-occurring species with smaller or shorter leaves (Scarff and Westoby 2006; Kane *et al.* 2008; de Maghalhães and Schwilk 2012). Furthermore, these species rated high across several flammability metrics because ignition was rapid, the flames produced were hotter and taller and the litter burnt out quickly (de Maghalhães and Schwilk 2012). Leaf structure is considered to be one of the most important morphological traits for determining flammability since leaves are usually the first structures to ignite and spread the fire to other fuel sources (Gill and Moore 1996; Etlinger and Beall 2004).

Flammability in forests refers to the capacity of plant materials to ignite and combust (Varner *et al*. 2015). There are complex interactions between litter fuelbed properties, chemical properties, physical properties and flammability measurements (Figure 1.1; Varner *et al.* 2015). Physical and chemical traits of litter drive flammability but, because there have been so few studies with common methods, terminology or comparable laboratory results or field observations of fire behaviour, studies are difficult to compare (Varner *et al*. 2015). It is crucial to quantify patterns and mechanisms of litter flammability for fire behaviour to be predicted in fire-prone ecosystems (Pausas and Moreira 2012). The majority of research has been in laboratories since field studies during bushfires are problematic because of complex interactions between topography, weather and fuels (e.g. Hiers *et al*. 2009; Loudermilk *et al*. 2012). However, the greatest challenge is to translate laboratory results to field-based observations of fire behaviour (Varner *et al*. 2015).

Realistic laboratory experiments testing flammability of forest litter and associated fire behaviour are lacking. Surface fuels consist of forest litter from dominant canopy species and understory plants (Hines *et al.* 2010; Gould *et al.* 2011; Berg and McClaugherty 2013) but litter flammability research has focused on single plant species in laboratories or altered litter samples for calorimetry or combustible gas analysis (Varner *et al*. 2015). Ignitability is usually tested on leaves or fuelbeds by ignition with a pilot heat source (Ganteaume *et al*. 2010). However, analysing ignitability is difficult because it is a binary response variable measuring multivariate flammability responses so correlating time to ignition with a litter component or combination of components in mixtures is obscured (Varner *et al*. 2015). Flammability metrics that have been quantified in laboratories include flame dimensions, flaming duration, duration of smoldering combustion, residual ash as a metric of fuel consumption, mass loss rates (Kreye *et al*. 2011), rates of spread (Ormeño *et al*. 2009; de Maghalhães and Schwilk 2012), and temperatures above a fuel bed measured with thermocouples (e.g. Santana and Marrs 2014; Varner *et al*. 2015). Flammability metrics tend to be consistent among studies, but methods are not and there can be intraspecific differences with the same litter type (e.g. Fonda 2001; Fonda *et al*. 1998). Quantifying the multivariate responses of flammability remains a challenge (Kremens *et al*. 2010).

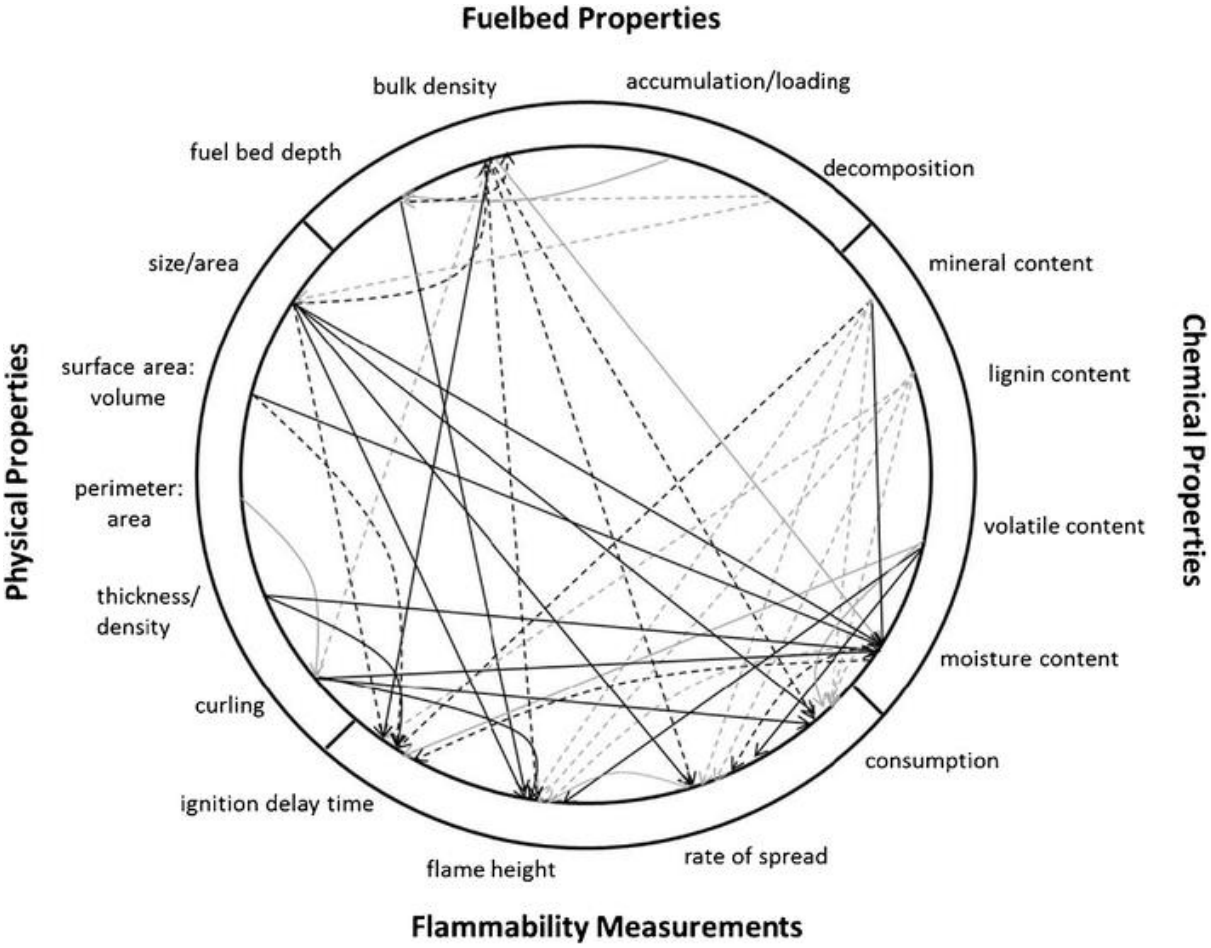

Figure 1.1 Relationships among physical properties, chemical properties, fuelbed properties, and flammability measurements. *Black lines* represent relationships based on published lab-based studies of litter flammability, and *grey lines* represent possible relationships based on other studies. *Solid lines* refer to positive relationships and *dashed lines* refer to negative relationships (Source: Varner *et al*. 2015).

Table 1.8 Leaf lengths of plant species from flammability studies in Hornsby Shire, Australia; California, USA; Florence, Italy and Guadalajara, Spain. (Adapted from: this study; Fairley and Moore 2010; de Maghalães and Schwilk 2012; Della Rocca *et al.* 2018). BD – Bulk density.

Plant species	**Leaf length (cm)**	**Longest length (cm)**	**Lowest BD**	**Study site**
Eucalyptus haemastoma	Up to 15			Halls Creek, Rofe Park
Eucalyptus pipperita	Up to 18			Rofe Park
Angophora costata	10–15			Rofe Park
Banksia serrata	Up to 16			Halls Creek, Rofe Park
Allocasuaraina littoralis	Branchlets – 20 + Leaves up to 8	Branchlets + leaves - 28	Lowest	Rofe Park
Cerapetalum gummifera	Up to 8			Halls Creek, Rofe Park
Corymbia eximia	Up to 20			Halls Creek
Corymbia gummifera	Up to 20			Rofe Park
Leptospermum trinervium	Up to 0.5			Halls Creek
Pinus jeffreyi	12–23	23	Lowest	California
Pinus lambertiana	5–10	10	Lowest	California
Pinus ponderosa	10–23	23	Lowest	California
Abies concolor	1.5–6			California
Abies magnifica	2–3.5			California
Calocedrus decurrens	0.3–1.4			California
Quercus kelloggii	8–15	15	Lowest	California
Sequoindendron giganteum	0.3–0.6			California
Cupressus sempervirens var. *horizontalis*	0.2–0.5			Florence
Quercus ilex subsp. *ilex*	4–8	8	Lowest	Florence
Juniperus thurifera	0.06–0.3			Guadalajara
Quercus faginea	4–10	10	Lowest	Guadalajara

1.7 Summary

Many strategic assets in Hornsby Shire are at extreme risk of bushfires. This is typical of densely populated areas that are near the urban-bush interface in south-eastern Australia. Fire mitigation treatments are very expensive and resources are limited. The plant community in a forest determines the type of forest litter that will build up on the forest floor and become the fuel load that will be burnt during prescribed burns. The vegetation types in Hornsby Shire in the Sydney Basin are extensive, very diverse and include dry and wet sclerophyll forest, turpentine-ironbark forest, rainforests and heaths. Studies in Australia have concentrated on sclerophyllous *Eucalyptus*-dominated temperate forests but there is little research for most of the other vegetation types. Keith (2004) has provided broad classes of vegetation that land managers can use to assess the risk of bushfires to strategic assets. However, there may be differences in constituent plant species within these broad classes of vegetation that can make certain sites become more flammable. The plant community also influences the microbial communities in forests and this ultimately affects the amount of carbon and nitrogen sequestered in soil. The configuration of surface fuels in litter including fine fuel, decomposing leaves and coarse woody material will also affect carbon and nitrogen cycles and carbon emissions.

To determine the flammability of the components of the surface fuel bed in a typical vegetation type in Hornsby Shire, a comparative study was done. Broad vegetation classification systems cannot capture finer details related to specific sites and therefore cannot account for differences in flammability that may be related to different leaf morphology and chemistry. The research presented in this study will provide a method that land managers can use to aid in their decision-making processes with respect to prescribed burning and its associated goals.

Risk is defined as likelihood and consequence of bushfires in fire-prone environments, including vegetation and urban structures (Sharples 2009). The Australian Standard states that risk is the probability of an occurrence that will affect objectives (Standards Australia 2004). Land managers currently use Rural Fire Service guidelines, broad vegetation classes, and information about fuel loads, area burnt during fires, and local knowledge to quantify risk to assist with planning prescribed fire regimes. If managers know which plant species, or mixture of species, are more flammable than others then this will be useful information to draw on. However, patterns and mechanisms of litter flammability must be quantified with realistic laboratory experiments, so the results can be transferred to fire-prone ecosystems.

This study will address three questions:

1. Do two sites classified by Keith (2004) as Sydney Coastal Dry Sclerophyll Forest differ in structure and fire risk?

2. Do sites in the Hornsby Shire that are classified as the same vegetation type, Sydney Coastal Dry Sclerophyll Forest, differ in flammability?

3. Can information about fuel loads and flammability be used to guide land managers in mitigation of risk from bushfires?

This study aims to compare fuel loads, visual and chemical analyses of forest litter from different locations in Hornsby Shire in the Sydney Basin to provide information about how the different components of forest litter found in these forests may affect fuel flammability. Chapter 2 provides quantitative measurements of the physical properties of fuelbed properties as measured in two sites classified as Sydney Coastal Dry Sclerophyll Forest. Chapter 3 will provide flammability measurements of the same forest litter and will describe a methodology that land managers can use when assessing fire risk. Chapter 4 summarises the findings of this study and specifies future directions of research.

Chapter Two: Characterising and quantifying surface fuel layers of Sydney Coastal Dry Sclerophyll Forest

2.1 Introduction

2.1.1 Fuel layers in Australian vegetation

In Australian forests there are generally five distinct fuel layers that influence fire behaviour (Gould *et al.* 2011; Hollis *et al.* 2015). All of the layers have a mixture of live and dead materials. At the ground level, the surface fuel layer consists of organic material on the forest floor and is often referred to as the litter layer (Hines *et al.* 2010; Gould *et al.* 2011 and see Section 1.3.1). It may range from being absent to more than 3 cm deep depending on forest type and time since disturbance (Hines *et al.* 2010; McCaw *et al.* 2012). The next layer, the near-surface fuel layer, consists of understorey vegetation (i.e. grass, herbs, short shrubs, vines) and plant material that may have fallen onto this vegetation. This layer ranges from a few centimetres from the ground to more than a metre high (Gould *et al.* 2011). The elevated fuel layer consists of tall understorey species (i.e. tall shrubs and small trees), bark from senescing shrubs and regenerating overstorey species (Gould *et al.* 2011). The intermediate layer is a mixture of mid-storey species including short mature trees and juvenile canopy plants (Gould *et al.* 2011). Bark is a primary fire hazard in the intermediate and overstorey tree canopy layers (Gould *et al.* 2011). The overstorey or tree canopy layer consists of dominant and co-dominant trees and forms the highest forest canopy layer (Gould *et al.* 2011; Hollis *et al.* 2015).

Characterisation of the surface fuel layer is crucial because it is where fires most often start and spread to other fuel layers. In seasonally dry forest systems such as Sydney Coastal Dry Sclerophyll Forest, the bulk density of litter influences flammability and, since litter accumulates because of low rates of litter decomposition, this fuel layer can substantially increase fuel load (van Wagtendonk *et al.* 1998; Bradstock and Cohn 2002; Stephens *et al.* 2004; van Wagtendonk and Moore 2010). Quantifying this fuel type is important for understanding fire behaviour, fire effects and management actions (Riccardi *et al.* 2007; Hollis *et al.* 2015). Both flaming and smouldering combustion is supported by fine fuels which are often found in surface litter (Ottmar 2014; Gould *et al.* 2011). Fires involving forest litter can cause considerable damage (Luke and McArthur 1986; Johnson and Miyanishi 2001; O'Bryan 2005 a, b) because this layer provides most of the fuel for rapid propagation of fire (Gould *et al.* 2011). However, this is not always the case as prescribed fires can be managed to remove the litter layer and lower vegetation without disturbing the tree canopy and upper understorey.

Leaves in forest litter influence fire behaviour. Leaf structure is considered to be one of the most important plant morphological traits to describe fuel flammability as leaves are usually the first structures to ignite which allows fire to spread to other fuel sources (Gill and Moore 1996; Etlinger and Beall 2004). Leaf size affects bulk density of litter as larger leaves make the surface fuel bed less dense allowing greater oxygen availability and increased heat diffusion (Scarff and Westoby 2006; Kane *et al.* 2008; Schwilk and Caprio 2011). Previous studies have demonstrated that species with large or long and thin leaves ignite more rapidly, burn quicker with more heat release and have higher flames (Fonda 2001; Scarff and Westoby 2006; Kane *et al.* 2008).

Particle size, fuel structure and arrangement, and moisture content of forest litter affect fire behaviour. One of the earliest studies investigating fire behaviour was done by Rothermel (1972) with combustion of fuel in a wind tunnel demonstrating that fuel load, bulk density and fuel particle size can affect fire spread. The high surface area to volume ratio of forest litter decreases time to ignition and contributes to fast rates of spread (Chandler *et al.* 1983). A loosely compacted litter layer can contribute to flame height with large twigs and the upper layer of the fuel bed contributing to flame depth behind the flame front (Gould *et al.* 2011). Lower, compacted layers of the fuel bed and coarse woody material are more likely to contribute to smouldering combustion (Gould *et al.* 2011). High moisture content decreases flammability by increasing the thermal capacity of live and dead plant tissues thereby suppressing flaming combustion (Etlinger and Beall 2004).

Each component of litter can potentially dominate fire behaviour of the mixture. A non-additive effect occurs when the fuel type with the greatest fire characteristic values influences fire behaviour by surpassing the proportion of its mass in the mixture (van Altena *et al.* 2012; de Magalhães and Schwilk 2012; Della Rocca *et al.* 2018). For example, the most ignitable fuel type has a strong non-additive effect when it dominates flammability of the mixture (van Altena *et al.* 2012). This non-additive effect has been demonstrated in experiments involving two species mixtures (van Altena *et al.* 2012), litter beds consisting of fuel from one (monospecific) or three different plant species at a time (de Magalhães and Schwilk 2012) and litter beds consisting of two different species (Della Rocca *et al.* 2018).

2.1.2 Characterising fuel flammability

Using small samples of vegetation to evaluate thermal degradation and combustion attributes has been used extensively and is associated with established methodologies and test standards (White and Zipperer 2010). Oxygen bomb calorimetry and elemental analysis provide information about the influence of chemical composition on combustion characteristics but does not incorporate physical characteristics of plant litter that can also affect flammability (White and Zipperer 2010). Thermal analysis of litter is useful for fire models because heat exposure is defined as a rate of heating which is a function of temperature (White and Zipperer 2010). The total energy content released during pyrolysis is the gross heat value or higher heating value (HHV) (McKendry 2002). This includes the latent heat from water vapour so the maximum amount of energy that can be recovered from the biomass source is given. However, the latent heat from water vapour cannot be efficiently converted during thermal heating so the lower heating value (LHV) will be closer to the actual amount of energy that is recovered. In addition, carbon to nitrogen ratios (C:N) have long been recognised as useful indicators of the source of organic matter, the state of decomposition and the possible contribution to soil fertility (e.g. Alexander 1977; Swift *et al*. 1979; Paul 2007).

2.1.3 Surface litter on the urban-bush interface

Flammability of surface litter is an important fire management issue in locations with strategic assets. The urban-bush interface is where buildings and infrastructure abut bushland that represents highly combustible fuels (Collins 2005; Cottrell 2005; Lein and Stump 2009). This area is particularly difficult to manage since there are often multiple ownerships, including public and private land, which often have different land management objectives (Reinhardt *et al.* 2008). Preventing property loss due to fire while achieving sustainable management of ecosystems, ensuring maintenance of biodiversity and reducing carbon and greenhouse gas emissions as primary land management objectives can be difficult to achieve at the same time (Reinhardt *et al.* 2008; Boer *et al.* 2009; Bowman *et al.* 2009; Bradstock *et al.* 2012). Information about fuel loads and fuel flammability can be used to guide land managers in mitigation of risk from bushfires. However, there is a lack of research evaluating the flammability of litter using both physical and biochemical traits across different plant species even though both of these components individually and interactively have long been considered to be important for fire behaviour (Rundel 1981; Hollis *et al.* 2015; Varner *et al.* 2015).

2.1.4 Research hypothesis and aims

The main purpose of the research presented in this chapter is to compare the physical properties of vegetation (fuel) properties at two sites representative of long unburnt Sydney Coastal Dry Sclerophyll Forest. In particular, the composition of the surface litter will be measured. Sydney Coastal Dry Sclerophyll Forest are open eucalypt forests and woodlands that are 10–25 m tall having a conspicuous and varied sclerophyll understorey and sedges as groundcover (Benson and Howell 1994; Keith 2004). This forest type is found along the coastal plateaux of the Sydney Basin at elevations below 700 m and is unique to New South Wales (NSW) (Benson and Howell 1994; Keith 2004). The urban-bush interface associated with this forest type is extensive.

The aim of this study was to test whether two sites classified by Keith (2004) as Sydney Coastal Dry Sclerophyll Forest differed in structure and fire risk. The objectives are to: (i) characterise the vegetation (fuel layers), and (ii) quantify the composition of the surface litter layer in two sites representing Sydney Coastal Dry Sclerophyll Forest.

2.2 Materials and methods

2.2.1 Study sites and plot establishment

The two sites were located near Bay Road, Arcadia (33°40'42" S, 184°53'55" E; hereafter referred to as 'Halls Creek') and in Rofe Park (33°36'59" S, 184°55'46" E; hereafter referred to as 'Rofe Park') in NSW, Australia. Both sites are located on public land managed by the Hornsby Shire Council. Long-term maximum monthly temperatures over 30°C were experienced in summer months (December and January) and minimum temperatures of 4.5°C and 5.5°C were in the winter months of July and August (Figure 2.1a). The elevation of study sites ranges from 142–206 m above sea level and the general study area has a mean average annual precipitation of 562 mm to more than 2844 mm (see Figure 2.1b). This wide range in precipitation from semi-arid to humid is typical of the Sydney region (Hughes and Crawford 2013). Sydney has a temperate climate with the wettest months from January to March and moderate inter-annual rainfall variability (Hughes and Crawford 2013). One plot in the study site at Arcadia was last burned in planned fires in 1990 and one plot at Rofe Park was burnt in 1996 (pers. comm. Brown and Jones 2016).

According to the widely used classification by Keith (2004), the vegetation of the sites selected are classified as 'Sydney Coastal Dry Sclerophyll Forest' and are in the sub-categories of 'Bloodwood-Scribbly Gum Woodland' (Halls Creek, Arcadia) and 'Peppermint Angophora Forest' (Rofe Park) for local land management purposes (Smith and Smith 2008). Both sites varied in the mix of canopy species although, each site was dominated by Scribbly Gum (*Eucalyptus haemastoma*) (Fairley and Moore 2010).

Three plots, each a 50 m transect, were established within each study site. The slope of the site was measured with a clinometer at the mid-point of each transect and a compass was used to determine the bearing and aspect. A global positioning system with a Global Navigation Satellite System Receiver (Trimble Inc., Sunnyvale, CA, USA) was used to measure location and altitude for each site.

2.2.2 Characterisation of overstorey canopy

For estimation of tree density, the five trees nearest the mid-point of the transect (25 m) with diameters greater than 10 cm were identified and the distance from the mid-point and their diameters were measured at breast height (DBH) (Cottam and Curtis 1956). Tree density (D) was calculated using the 'nearest individual method':

$$D = \frac{1}{(\Sigma r/n)^2}$$

In this equation, r represents the distance (m) measured from a fixed point to each tree and n is the number of samples. The main overstorey tree species associated with each plot transect were recorded.

2.2.3 Vertical structure assessment

The pin transect method was used to assess the vertical structure of the fuel (Gould *et al.* 2011). At 5 m intervals along each transect (i.e. 0, 5, 10, 15, 20, 25, 30, 35, 40, 45 and 50 m), a 200 cm pole marked with intervals (0–20, 20–50, 50–100, 100–150 and 150–200 cm) was randomly placed on the ground near the measurement point and held vertically. For each height interval, any litter or vegetation that touched the pole was counted and recorded in the following categories; 'woody' (dead fuel >6 mm diameter), 'dead fine' (dead surface litter <6 mm diameter) and live fuels as 'grass', 'herb' or 'tree'. In addition, at each point along the transect, litter depth was recorded as the average of 10 randomly placed measurements.

For each transect, a visual assessment of the vertical fuel structure (Gould *et al.* 2011) was done at the 5, 15, 25, 35 and 45 m points along the transect. This method involves identification of five fuel layers: 'surface fuel', 'near-surface fuel', 'elevated fuel', 'intermediate tree canopy' and 'overstorey tree canopy'. A subjective rating, the 'fuel hazard score' (FHS) is given for each fuel layer. The FHS for the surface litter layer was from 0 to 4 according to litter depth: 0 indicates that litter is absent; 1 is <10 mm litter; 2 is 10–20 mm; 3 is 15–25 mm; 4 is >25 mm. The remaining fuel layers were described according to 'percent cover score' (PCS) according to an estimate of the amount of cover of each layer: 0 is no cover; 1 is 1–25% cover; 2 is 25–50%; 3 is 50–75%; 4 is >75% (Gould *et al.* 2011). 'Surface fuel' was ranked by the amount of litter covering the ground: 0 is 0–25% cover; 1 is 25–75%; 2 is 75–90%; 3 is 90-–95%; 4 is >95% (Gould *et al.* 2011). Near surface fuels are up to 1 m tall and consist of litter suspended on grass, short shrubs, vines and collapsed understory vegetation (Gould *et al.* 2011). Elevated fuels are up to 3 m tall and include live and dead vegetation of the understorey vegetation and regenerating trees (Gould *et al.* 2011). The height of near surface and elevated fuels were also measured. A PCS score was given for percentage of canopy cover.

Fuel hazard scores for the bark associated with the intermediate and overstorey tree canopies were ranked from low risk (0) to a high fuel hazard (4): 0 indicates there is no bark on the tree; 1 is for stringybarks where the bark is tightly held on the whole tree trunk and is well-charred or ironbarks with very tight, platy or fibrous bark or smooth barks; 2 is for well-charred stringybark on the lower trunk or a few pieces of loosely attached or tight fibrous bark that has not been burnt for a long time or a mixture of bark with long loose ribbons and smooth bark down to the ground; 3 is stringybark with <50% of the tree well-charred or if the upper part of the tree is not charred or smooth/candle barks with long loose ribbons of bark or fibrous or platy bark on the lower trunk that has not been burnt for a long time; 4 is large long loose flakes of bark all the way down the trunk from the branches to the ground or the bark is loosely attached to the trunk (Gould *et al.* 2011 and see Section 1.3.1).

2.2.4 Litter, live biomass and coarse woody debris

At five sample collection points along each transect (i.e. 5, 15, 25, 35 and 45 m), a circular sampling ring with 35.5 cm diameter (0.1 m^2) was placed on the ground and the live vegetation within the ring was clipped and collected. The litter depth (cm) was measured at six points within the ring. After the live vegetation had been removed, all of the litter within the ring was collected down to the mineral soil layer. Care was taken to collect only the litter but, in some instances, mineral soil was also collected because it had been mixed into the litter. Temperature (QM7217, Digitech, China) and soil moisture probes (MPM-160-B, ICT International Pty Ltd, Armidale NSW Australia) were used to take six measurements within each collection point.

Using the three 50 m transects established at each site, the line intersect method described by van Wagner (1982) was used to measure the quantity of coarse woody debris (CWD) on or near the ground. Coarse woody debris is defined as woody material with a diameter greater than 6 mm (van Wagner 1982; Gould *et al.* 2011). There were 239 measurements of CWD made at Rofe Park and 390 at the Halls Creek site.

Live biomass and litter fractionation

Live plant material and litter was oven-dried at 60°C for 48 h and weighed. Dried litter samples were sorted into separate components and reweighed. Twigs with a diameter of 6 mm or less were included as litter. ‘Leaves’, ‘twigs’, ‘other’ composed of bark, woody fruits and other materials and ‘*Allocasuarina littoralis*’ greater than 9 mm were removed from decomposed materials of less than 9 mm with a sieve. The ‘decomposed’ fraction composed of particles of leaves, bark, small fruits and seed; leaves; twigs <9 mm. Soil was removed from decomposed materials with a sieve aperture of 2 mm. Four litter fractions were identified for Halls Creek (i.e. leaves, twigs, decomposed material and 'other' types of material such as woody fruits and bark) but the presence of *Allocasuarina littoralis* at Rofe Park prompted the inclusion of an additional fraction (i.e. *Allocasuarina* litter) for this site.

Correction factors for inorganic content of litter

To determine how much of the decomposed litter was organic and not mineral soil, it was necessary to calculate the inorganic content by heating representative samples in a muffle furnace. Ground samples were oven-dried to 105°C to remove moisture then heated to 250°C to avoid flaming before heating at 575°C to determine the amount of inorganic material remaining (American Society for Testing and Materials International Standard E1755-01; 2015). The inorganic content consisted of silica and ash (<1%). Thus, the amount of soil in the biomass samples (Table 2.2 and Table 2.3, respectively) could be corrected for and incorporated in calculations for determining chemical composition (Table 2.4). The proportion of silica was calculated as follows:

1. Calculate the oven dry weight (ODW) of each sample (values provided in Tables 2.2 and 2.3, respectively):

$$\mathrm{ODW} = \frac{\mathrm{Weight}_{\mathrm{air\ dry\ sample}} \times \%\mathrm{Total\ solids}}{100}$$

2. Calculate and record the ash percentage on an ODW basis

$$\% \ ash = \left[\frac{(m_{ash} - m_{cont})}{(m_{od} - m_{cont})}\right] \times 100$$

where: *% ash* is the mass percent of ash (based on 105ºC oven-dried mass of the sample), m_{ash} is the mass of ash plus porcelain crucible (g), m_{cont} is the tare mass of crucible (g) and m_{od} is the initial mass of the 105ºC oven-dried sample and crucible (g).

Since most of the inorganic material was silica rather than ash the column names refer to 'inorganic' (Table 2.2 and Table 2.3, respectively).

2.2.5 Carbon, nitrogen, hydrogen, C:N ratio and heat of combustion

Approximately 3 g of composite litter samples (less than 9 mm; n = 15 samples per site) were finely ground (Jaw Crusher SM100, Retsch 100 Grinding Mill, Haan, Germany) in preparation for chemical analysis. Elemental analysis (TruSpec Micro LECO, Saint Joseph, MI, USA) was used for determining the proportion of carbon, nitrogen and hydrogen in each sample. Values were adjusted to account for the presence of mineral soil (see Section 2.2.4) and C:N ratios were calculated. Oxygen bomb calorimetry (6400 Automatic Isoperibol Calorimeter; Parr Instrument Company, Moline IL) was used to determine the gross heat of combustion (n = 15 samples per site).

2.2.6 Statistical analysis

The Mann-Whitney-Wilcoxon test with parameters for the Mann-Whitney test (MW) from the R program, was used to compare differences among variables: tree density (trees ha^{-1}), surface depth (mm), near surface height (m), elevated height (m), fuel hazard scores (FHS) for surface, near surface, elevated and bark fuels, percent cover score (PCS) for surface, near surface, elevated and canopy fuels, pin transect data for grass, herb, woody, tree and dead fine at 0–20, 20–50, 50–100, 100–150 and 150–200 m intervals. A significance value of p <0.05 was used throughout.

2.3 Results

2.3.1 Study site characteristics

At the study site at Rofe Park, dominant overstorey and mid-storey tree species included *Eucalyptus haemastoma*, *E. pipperita*, *Angophora costata*, *Banksia serrata*, *Allocasuarina littoralis*, *Cerapetalum gummifera* and *Corymbia gummifera*. Dominant canopy and mid-storey species at Halls Creek included *E. haemastoma*, *Corymbia eximia*, *B. serrata*, *Leptospermum trinervium* and *Cerapetalum gummifera*. The average tree density at Halls Creek was 1358 ± 811 trees ha^{-1} and 1198 ± 400 trees ha^{-1} at Rofe Park with no significant difference between sites (p >0.05). Other measurements are presented in Table 2.1. For example, soil temperature at Halls Creek was 17.1 ± 0.4°C and 17.7 ± 1.2°C at Rofe Park with no significant difference between sites (p >0.05).

There was no significant difference in litter biomass between Halls Creek (7.2 ± 2.9 t ha^{-1}) and Rofe Park (8.6 ± 4.7 t ha^{-1}). In contrast, there were significant differences between the two sites for soil moisture at Halls Creek (3.5 ± 0.1%) and at Rofe Park (5.7 ± 1.3%) (MW = 6650, p <0.001), live biomass at Halls Creek (22.3 ± 6.1 t ha^{-1}) and at Rofe Park (35.2 ± 17.1 t ha^{-1}) (MW = 178, p = 0.006) and litter bulk density at Halls Creek (20.0 ± 5.3 t ha^{-1}) and at Rofe Park (25.7 ± 10.3 t ha^{-1}) (MW = 35, p <0.001).

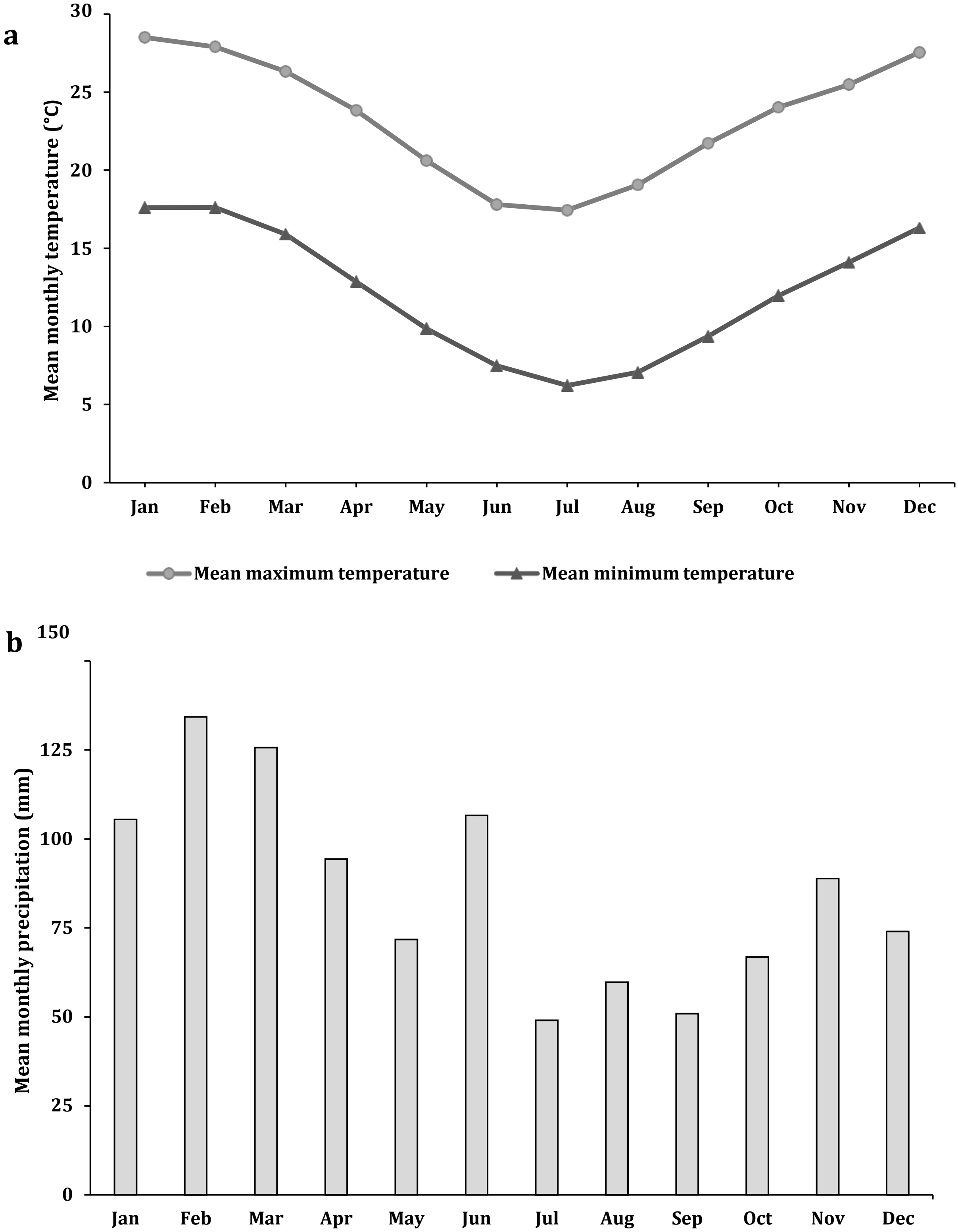

Figure 2.1 Long-term weather conditions describing the two study sites including (a) mean monthly temperature range of both sites from the closest weather station (North Parramatta, 25 km from sites, data from 1967–2018), and (b) mean monthly precipitation from Dural, the closest rain gauge to Halls Creek

and Rofe (11 and 13 km from site, respectively, data from 1973–2018) (Source: http://www.bom.gov.au/climate/data/index.shtml Access date: 12 September 2018).

Table 2.1 Site characterisation data. Different letters indicate significant statistical difference between sites.

Site characteristic	**Rofe Park**	**Halls Creek**
Soil temperature (°C)	17.7 ± 1.2[a]	17.1 ± 0.4[a]
Soil moisture (%)	5.7 ± 1.3[A]	3.5 ± 0.1[B]
Live biomass (t ha^{-1})	35.2 ± 17.1[*a*]	22.3 ± 6.1[*b*]
Litter biomass (t ha^{-1})	8.6 ± 4.7[*A*]	7.2 ± 2.9[*A*]
Litter bulk density (kg m^{-3})	25.7 ± 10.3[a]	20.0 ± 5.3[b]
Coarse woody debris (sound wood) (t ha^{-1})	0.42 ± 0.35[A]	0.97 ± 0.74[A]
Coarse woody debris (rotten wood) (t ha^{-1})	0.32 ± 0.27[*a*]	0.39 ± 0.30[*a*]

2.3.2 Vertical structure

According to the pin transect data, both sites had some type of fuel recorded at all heights and in all categories with the most touches by dead fine fuel in the 0–20 cm height interval (MW = 392, p = 0.047) (Figure 2.2). In the 0–20 cm height interval, the number of touches by grass (MW = 685, p = 0.043), herbaceous plants (MW = 394, p = 0.009), dead fine fuels (MW = 392, p = 0.047) were all significantly different at Rofe Park compared to Halls Creek. Both sites were similar for number of touches by woody material in the 0–20 cm height interval. For the 20–50 cm height interval dead fine fuels were significantly different between sites (MW = 445, p = 0.011). For the 50–100 cm height interval only herbaceous plants were significantly different (MW = 479, p = 0.042), and both sites were not significantly different for the number of touches by all fuel types in the 100–200 cm interval.

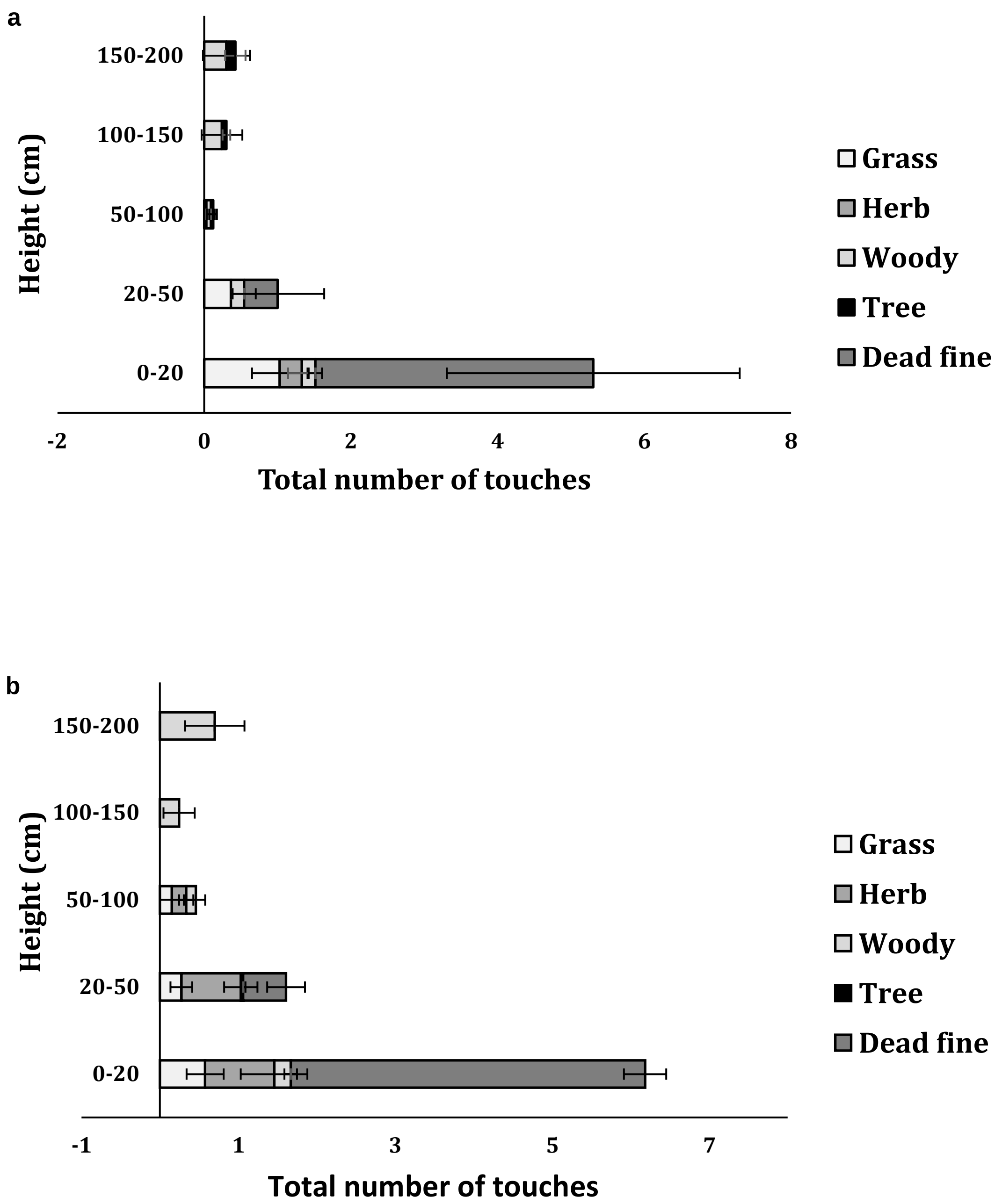

Figure 2.2 The pin transect method (adapted from Gould *et al.* 2011) was used to assess vertical fuel structure at (a) Rofe Park and (b) Halls Creek**.**

Litter depth at Rofe Park was significantly greater (75 ± 31 mm) than at Halls Creek (46 ± 17 mm) (Figure 2.3a; MW = 162, $p = 0.042$). Near surface fuel height was significantly greater at Rofe Park (1.0 ± 0.2 m) than at Halls Creek (0.4 ± 0.1 m; MW = 120, $p < 0.001$), and elevated fuel height was lower at Rofe Park (2.4 ± 0.7 m) than at Halls Creek (2.9 ± 0.3 m) but was not significantly different ($p > 0.05$; Figure 2.3b).

The greatest FHS was recorded for surface fuel at Rofe Park (3.7 ± 0.5) and was significantly greater than at Halls Creek (3.3 ± 0.5; MW = 168, $p = 0.006$) (Figure 2.3c). Fuel hazard scores for near surface and elevated fuels were also greater at Rofe Park than at Halls Creek but were not significantly different ($p > 0.05$). However, the FHS for bark was significantly lower at Rofe Park than at Halls Creek (MW = 75, $p = 0.018$).

The highest mean PCS was assigned to surface litter and was greater at Rofe Park (3.4 ± 0.2) than at Halls Creek (3.1 ± 0.4) but was not significantly different ($p > 0.05$; Figure 2.3d). The PCS for near surface and elevated fuels were also greater at Rofe Park than at Halls Creek but were also not significantly different. However, the PCS for canopy fuels was significantly lower at Rofe Park than at Halls Creek (MW = 184, $p < 0.001$).

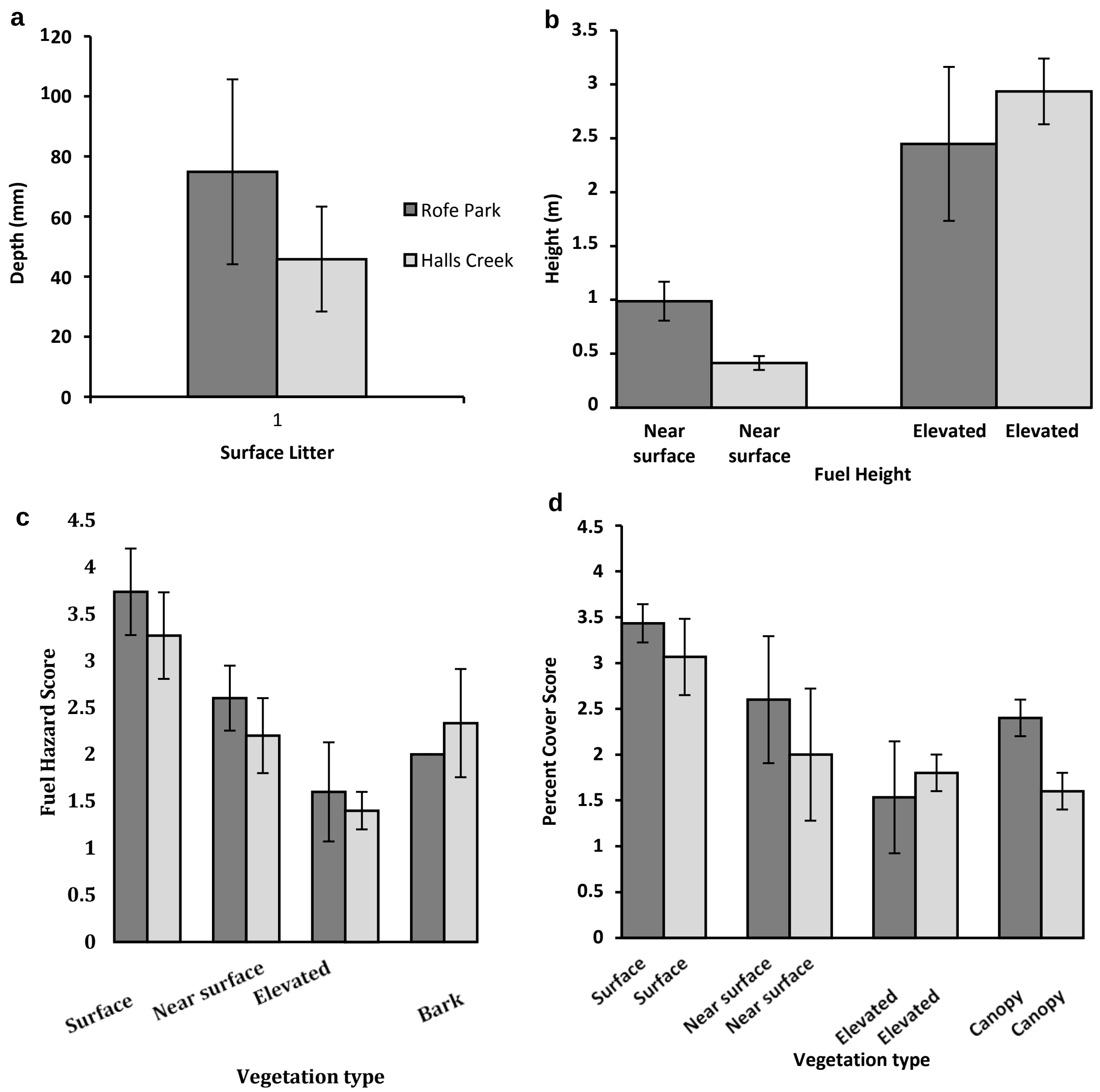

Figure 2.3 Physical measurements and visual assessments of fuel from Rofe Park and Halls Creek: (a) surface litter depth (mm), (b) fuel height (m), (c) fuel hazard score and (d) percent cover score. Bars are mean values and error bars are standard deviation.

2.3.3 Litter characteristics

The inorganic component of the decomposed fraction of the litter was lower for samples from Halls Creek (7.14 ± 5.55; Table 2.2) compared to Halls Creek (8.92 ± 12.64; Table 2.3). These values were used to adjust biomass calculations as described in Section 2.2.4.

Table 2.2 Proportion of inorganic material (silica) in the decomposed litter fraction from Halls Creek. Ground samples were oven-dried at 105°C and at 250°C before heating to 575°C for 3 h. Halls Creek 1, 2 or 3 refer to the site and transect. Multiples of five (m) are the position on the transect. St. dev. – standard deviation.

Sample ID	Crucible mass (g)	Dried sample + crucible mass (g)	Inorganic + crucible mass (g)	Inorganic material (%)
Halls Creek 1				
5 m	17.73	18.37	17.78	7.81
15 m	17.48	18.00	17.55	13.46
25 m	17.89	18.83	18.01	12.77
45 m	16.76	17.67	16.82	6.59
Halls Creek 2				
15 m	16.99	17.66	17.10	16.42
25 m	17.20	18.13	17.26	6.45
35 m	17.65	18.54	17.67	2.25
Halls Creek 3				
5 m	17.92	18.85	17.92	0
25 m	19.62	20.53	19.63	1.10
45 m	18.65	19.53	18.69	4.55
Mean				7.14
St. dev.				5.55

Table 2.3 Proportion of inorganic material (silica) in the decomposed litter fraction from Rofe Park. Ground samples were oven-dried at 105°C and at 250°C before heating to 575 °C for 3 h. Rofe Park 1, 2 or 3 refer to the site and transect. Multiples of five (m) the position on the transect. St. dev. – standard deviation.

Sample ID	**Crucible mass (g)**	**Dried sample + crucible mass (g)**	**Inorganic + crucible mass (g)**	**Inorganic material (%)**
Rofe 1				
15 m	17.43	18.66	17.46	2.44
25 m	14.06	14.60	14.08	3.70
45 m	17.09	18.00	17.13	4.40
Rofe 2				
5 m	17.49	18.92	18.10	42.66
25 m	15.99	16.94	16.01	2.11
35 m	18.77	19.52	18.79	2.67
Rofe 3				
5 m	18.59	19.52	18.63	4.30
15 m	18.01	18.74	18.13	16.44
35 m	17.01	17.93	17.03	2.17
45 m	17.71	18.55	17.78	8.33
Average				8.92
St. dev.				12.64

Litter from *Allocasuarina littoralis* ('Casuarina') was only found at Rofe Park (Figure 2.4). This type of litter is characterised by long thin needle-like cladodes (Figure 2.5). Biomass of litter, twigs and 'other' were similar for both Rofe Park and Halls Creek (p >0.05), but biomass of the leaf (MW = 54, p = 0.014) and decomposed fractions (MW = 206, p <0.001) were significantly different between the two sites.

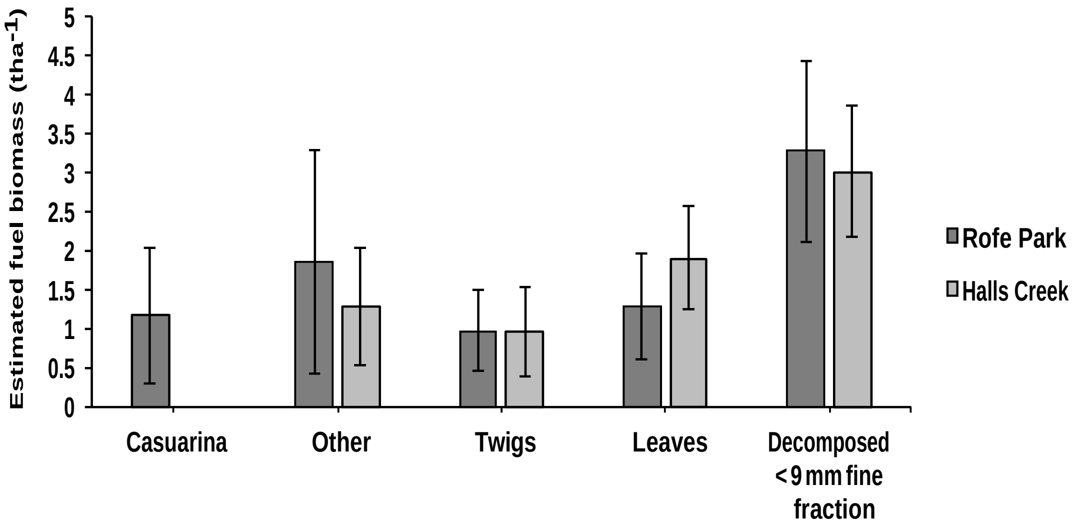

Figure 2.4 Biomass (t ha^{-1}) of five litter fractions from Rofe Park and four litter fractions from Halls Creek. Bars are mean values and error bars are standard deviation.

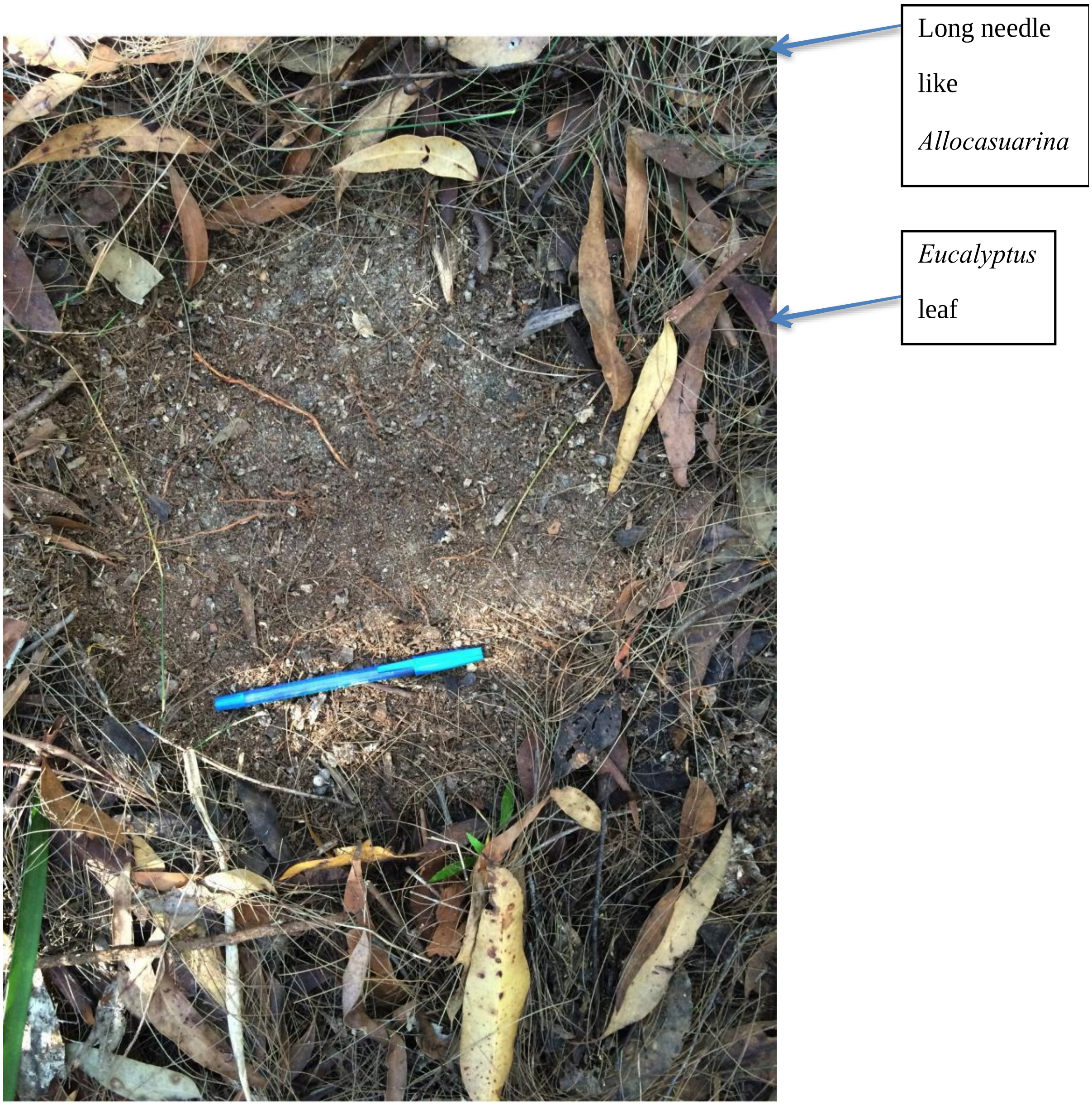

Figure 2.5 Litter of *Allocasuarina littoralis* cladodes mixed with *Eucalyptus* leaves from Rofe Park.

2.3.4 Chemical composition of litter

The carbon, nitrogen, and hydrogen content of composite litter (including all fractions) from Rofe Park and Halls Creek, along with related net heat of combustion (NHC) and gross heat of combustion (GHC), are provided in Table 2.4. The nitrogen values at Rofe Park were higher than at Halls Creek because *Allocasuarina littoralis*, a nitrogen-fixing plant, was only found at this site. The carbon values at both sites were similar. There was no significant difference in the NHC (p >0.05) or in GHC (p >0.05) of samples from Halls Creek and Rofe Park.

Table 2.4 Chemical and combustion analysis of composite litter samples <9 mm from Halls Creek and Rofe Park. Values (n = 15 per site) are mean ± standard deviation. GHC – gross heat of combustion; NHC – net heat of combustion.

Characteristic	**Halls Creek**	**Rofe Park**
Nitrogen (%)	0.63 ± 2.47	0.88 ± 0.15
Carbon (%)	43.92 ± 1.03	41.81 ± 1.25
Hydrogen (%)	6.27 ± 0.20	5.87 ± 0.20
C:N	69.71 ± 0.42	47.51 ± 8.33
GHC (MJ kg^{-1})	17.12 ± 0.20	16.42 ± 0.54
NHC (MJ kg^{-1})	14.74 ± 0.44	14.05 ± 0.51

2.4 Discussion

For dry sclerophyll forests in south-eastern Australia it is crucial to quantify production of vegetation, fuel accumulation (Specht and Specht 1999) and decomposition (Raison *et al.* 1986) and to determine whether understory herbs and shrubs are available to act as ladder fuel (Gould *et al.* 2011). The dominant tree species control the type of forest litter as a function of the proportion of aboveground biomass held in these individuals (Bradstock 2010). Both sites investigated had litter depths of up to 50 mm and, according to visual assessments, the fire hazard risk was in the category of extreme risk (Gould *et al*. 2011). This is consistent with the assessment of extreme risk of bushfires in the Overall Fuel Hazards Guide (Hines *et al.* 2010), with a very thick layer (>35 mm) of completely connected (continuous) litter. Despite similar risk ratings, surface litter depth and near-surface fuel height were greater at Rofe Park than Halls Creek. Similarly, the highest FHS were for surface litter fuels at Rofe Park and these were significantly different from Halls Creek. Near-surface fuel hazard score and near-surface height are both tightly correlated with rate of spread (Gould *et al.* 2008) which suggests that fire in Rofe Park would have a greater rate of spread than at Halls Creek.

Connectivity of vertical layers of vegetation enhances fire propagation (Peters *et al.* 2004; Pueyo 2007). Grasses and herbaceous materials are often considered to be ladder fuels between surface litter and trees (Hodgkinson 2002; Prober *et al.* 2007, 2008). Both of the sites investigated had similar vertical fuel structure as indicated by the pin point method but generally differed in composition. For example, there was more grass in the surface and near surface layers at Rofe Park than at Halls Creek, and more herbs at Halls Creek. Studies have found continuity of the near-surface fuel layer with suspended litter on live vegetation, such as shrubs, herbs and grasses, significantly affect rate of fire spread (Burrows 1994; Cheney *et al.* 1992; McCaw *et al.* 2008; Burrows 2013).

The elevated and intermediate tree and canopy layers are also important in spreading fire into the canopy (Gould *et al.* 2011). The vertical fuel structure of both sites indicated a risk of fires spreading into the canopy. This is typical of unburnt dry sclerophyll forests since a defining feature is a conspicuous shrub layer under trees with open canopies (Keith 2004). These forests have an especially diverse range of vegetation that include sclerophyllous plants and sedges in a shrubby sub-formation whereas the shrub/grass sub-formation of dry sclerophyll forest has a mixture of sclerophyllous and non-sclerophyllous vegetation with a dominance of grasses (Keith 2004).

Litter depth and near surface fuel height have a profound effect on flammability and are often considered to be the most important fuel variables affecting fire behaviour in eucalypt forests (McArthur 1962; Peet 1965; McArthur 1967). Studies have found a correlation between rate of spread and fuel load for low intensity fires (<2500 kW m^{-1}), but there is little evidence of this with higher intensity fires (Cheney 1990; Burrows 1994, 1999).

Bark condition, particularly when it is abundant and fibrous, is well-known to contribute to flame height (Gould *et al.* 2008). This fuel type can be described by intermediate and overstorey fuel hazard scores. Information about the amount of bark in an area is important as it can contribute to spot fires that breach barriers in fuel, including roads and fire breaks, and enable fires to be rapidly transmitted down slopes (Gould *et al.* 2008). Large overstorey tree species determine the flammability of several vertical fuel layers depending on bark characteristics (Gould *et al*. 2011). Bark can contribute to surface fuel depth, spreading fire from this layer into the canopy and generating firebrands (Gould *et al.* 2011). There were a number of dominant canopy and mid-storey tree species at Halls Creek, each with different bark features. For example, *Eucalyptus haemastoma* has smooth patchy white to yellowish or greyish bark and was generally associated with a low FHS. *Corymbia eximia* has rough tessellated flaky yellow-brown bark extending to smaller branches and also scored a low FHS. In contrast, *Leptospermum trinervium* has a high FHS as it has rough flaky bark (Hornsby Shire Council 2018a, b; Fairley and Moore 2010; Gould *et al.* 2011). At Rofe Park, dominant overstorey and mid-storey tree species included species with both low (i.e. *E. haemastoma*) and high FHS (e.g. *E. pipperita* with short fibrous grey bark on the trunk and larger branches and ribbony bark on upper branches, *Angophora costata* which sheds bark in patches). (Fairley and Moore 2010; Gould *et al.* 2011; Hornsby Shire Council 2018a, b). Both sites had similar bark flammability risk despite having different species composition with different bark and flammability qualities.

An abundance of *Allocasuarina littoralis* litter at Rofe Park greatly increased the amount and bulk density of surface litter and near surface fuel. Apart from adding to the fuel load, a thick dense layer of *Allocasuarina* litter may also promote warmer and wetter soils. This hypothesis is somewhat supported by site characterisation as soil moisture at Rofe Park on the day of sampling was greater than at Halls Creek. Soil temperature and moisture content affect decomposition rates of litter with warmer and wetter conditions increasing the rate of activity (Davidson and Janssens 2006). As expected from this pattern, Rofe Park has more decomposed material than at Halls

Creek. It must be remembered that there are many other variables that can affect the amount of decomposed material present on the forest floor (Davidson and Janssens 2006), so the site characterisation done in this study is only an indication of potential difference in the ability of the two sites to burn.

Surface litter from both sites had similar values for GHC (16.4–17.1 MJ kg^{-1}) and NHC (14.0–14.7 MJ kg^{-1}) but differed according to C:N ratio (ranging from 48–70). The proportion of nitrogen and C:N ratios (see Tables A5 and A6 in the Appendix) are comparable to the experimental values of Snowdon *et al.* (2005) (see Section 1.1, Table 1.3) for Eucalypt Coastal Woodlands, the type of vegetation represented at the study sites. In general, a C:N ratio higher than 25, on a mass basis, demonstrates that decomposition is slower than the rate of accumulation of litter (Bui and Henderson 2013), as was the case at both of the study sites. A C:N ratio lower than 25 indicates that the forest is productive (Snowdon *et al.* 2005). Leaching can cause the C:N ratio of decomposing litter to change (Snowdon *et al.* 2005), as can fungal and bacterial activity (Hodge *et al*. 2000; Makino *et al*. 2003) causing high C:N ratio values, such as that measured for litter from Rofe Park. In addition, C:N ratios are lower for nitrogen-fixing species such as *Allocasuarina littoralis* than for non-nitrogen-fixing species (Snowdon *et al.* 2005). While information about C:N ratio of litter can provide useful information about the condition of a forest site and some of the turnover of plant material that is occurring, it is not a strong indicator of how flammable a site may be.

Allocasuarina littoralis was abundant at Rofe Park and provided fuel connectivity with surface, near-surface and elevated fuels. How the flammability of a landscape is changed by a different species is poorly understood (Bradstock 2010), although it is known there can be intraspecific differences in vegetation within the same forest type that affect flammability (Fonda 2001; Fonda *et al*. 1998). For example, pines that are 'fire resisters' will burn rapidly whereas 'fire evaders' tend to burn slowly (Fonda 2001; Fonda *et al*. 1998). This is consistent with the finding of leaves rapidly igniting with great intensity having a high surface area to mass ratio (Schwilk 2015).

Similarly, Cornwell *et al.* (2015) found that different litter flammability within the family Pinaceae is determined by the size of abscised needles and branches. Different flammability traits existed in the Cretaceous and, as surface fires were important during this period (Bond and Scott 2010; He *et al.* 2012), the implication is that fire regimes were affected by species effects on litter flammability (Schwilk 2015). A range of studies have also demonstrated that larger litter particle size influences flammability with more aeration, more rapid rate of spread, and higher heat release rates (Scarff and Westoby 2006; Kane *et al.* 2008; de Magalhães and Schwilk 2012; van Altena *et al.* 2012).

Determining how forest litter influences flammability and fire behaviour in temperate forests is an important problem. There have been many studies attempting to measure flammability of vegetation including field burning experiments (Grant *et al.* 1997; Ellair and Platt 2013), burning tables or benches (Morandini *et al.* 2013; Ormeno *et al.*, 2009; Santoni *et al.* 2011; Ganteaume *et al.* 2010) and testing individual leaves (Gill and Moore 1996; Engstrom *et al.* 2004). However, the extent that fuel load influences fire behaviour continues to be debated (Mutch 1970; Troumbis and Trabaud 1989; Bond and Midgley 1995; Schwilk 2003; Fernandes and Cruz 2012; Pausas and Moreira 2012) mainly because there have been discrepancies between laboratory-scale testing, field results and modelling outputs (Fernandes and Cruz 2012). It is crucial to find a mechanistic basis to scale from laboratory flammability testing of plant traits to fire behaviour in the field, but it has remained elusive (Schwilk 2015).

Knowledge of where fuel comes from and their physical and chemical properties are important to document and can be used for interpretation of flammability, but they are limited in actually predicting or understanding fuel flammability. In this study, both the Halls Creek and Rofe Park sites have been described as the ‘context’ for litter fuel and a few flammability measures (gross heat of combustion, NHC, C:N ratio) have been presented. In Chapter 3, flammability measurements of litter from both sites will be investigated.

Chapter Three: Flammability of litter from Sydney Coastal Dry Sclerophyll Forest

3.1 Introduction

Empirical data involving the patterns and mechanisms of forest litter flammability in an ecological context are needed to assist with land management of fire-prone ecosystems (Varner *et al*. 2015). Despite the importance of understanding litter flammability many aspects are unknown. There are no standards or test procedures to determine the flammability components of plants (White and Zipperer 2010). This study aims to fill this research gap by using the flammability metrics of forest litter from areas of extreme risk of bushfire to add to the existing physical measurements that land managers use, such as bulk density, to prioritise prescribed burns in sites with vegetation of known risk in terms of conditions at which there is a change in fire behaviour (known as 'ignition thresholds'). Plucinski and Anderson (2008) apply this term to bushfires that cause a sustainable fire spread beyond the immediate influence of the pilot ignition source. Ignition thresholds provide important information for planning and implementing prescribed fire and for determining fire danger. Such information can be used in conjunction with fire behaviour models so that fire spread is only predicted when the vegetation will burn (Plucinski and Anderson 2008).

3.1.1 Methodology rationale

The purpose of this study was to compare the flammability of litter from two sites from the same class of vegetation in south-east Australia. Flammability of vegetation has four components; ignitability, combustability, sustainability and consumability (White and Zipperer 2010). Ignitability has two definitions; the minimum temperature or heat flux required for ignition, or how quickly plant litter will ignite after exposure to an external heat source (Anderson 1970; Gill and Moore 1996; Dimitrakopoulos and Papaioannou 2001; Behm *et al.* 2004). Combustibility refers to how rapidly a fire burns (Anderson 1970; Behm *et al.* 2004). Suitable measurements include heat release, temperature and visual observations of flames (White and Zipperer 2010). Sustainability determines the ability of the fire to continue to burn regardless of whether there is a heat source (Anderson 1970; Mak 1988; Behm *et al.* 2004). Heat of combustion and total heat release are useful indicators of sustainability (White and Zipperer 2010). Consumability determines the amount of plant litter consumed by the fire (Martin *et al.* 1994; Behm *et al.* 2004). When ignition does not occur, the other three components become irrelevant (Martin *et al.* 1994) so, in this study, flammability measurements were taken if the sample ignited (Table 3.1).

Table 3.1 Flammability components, their definitions and examples of fire test measurements. (Adapted from: White and Zipperer 2010). TTI – time to ignition; VFH – visual flame height; RS – rate of spread; BTC – burn to completion; RMF – residual mass fraction; VC – volume consumed; DVF – duration of visual flaming.

Components	Definition	Potential test responses	Metric
Ignitability	Time until ignition once exposed to a heat source	Fuel ignited (Y/N) Ignition time (s)	TTI
Combustability	Rapidity of combustion after ignition	Visual flame height (m) Rate of spread (m^{-1})	VFH RS
Consumability	Proportion of mass or volume consumed by combustion	Burn to completion (s) Fuel burn to edge (Y/N) Mass loss rate Fuel mass (%) after burning	BTC VC RMF
Sustainability	Ability to sustain combustion once ignited	Duration of visual flaming (s) Area or volume consumed (m^2, m^3)	DVF

To determine which component or mixture of litter is the most flammable a Simplex Centroid Design (SCD) was used. This design method is a multivariate design of experiment (DOE) that is commonly used for analytical chemistry experiments when optimisation is an essential stage to determine the value that each factor must provide to ensure an optimal outcome (Candioti *et al.* 2014). The advantages of using a multivariate DOE, instead of univariate procedures, is that it is more efficient, requires less materials, and provides a great deal of information while reducing the amount of experiments needed for multiple response optimisation (Candioti *et al.* 2014). In addition, multivariate DOE varies all the levels of the factors involved simultaneously which enables a mathematical model to be created to connect the response to the experimental conditions. These responses at any point of the experimental domain can be predicted once the coefficients of the model have been estimated (Candioti *et al.* 2014). Furthermore, the interactions between the factors with the responses can be studied (Candioti *et al.* 2014).

Litter fractions are not generally independent from each other because they are mixed together in the field and are collected as a composite sample. A mixture design that allows the variation of the ratios among the litter fractions is necessary. The SCD enables this to occur since the experimental design has a domain as a regular figure with as many vertices as components and a space with dimensionality equal to the number of components minus one (Leardi 2009). For example, the representation of a three-component mixture is an equilateral triangle with three vertices presented in a two-dimensional space (Figure 3.1).

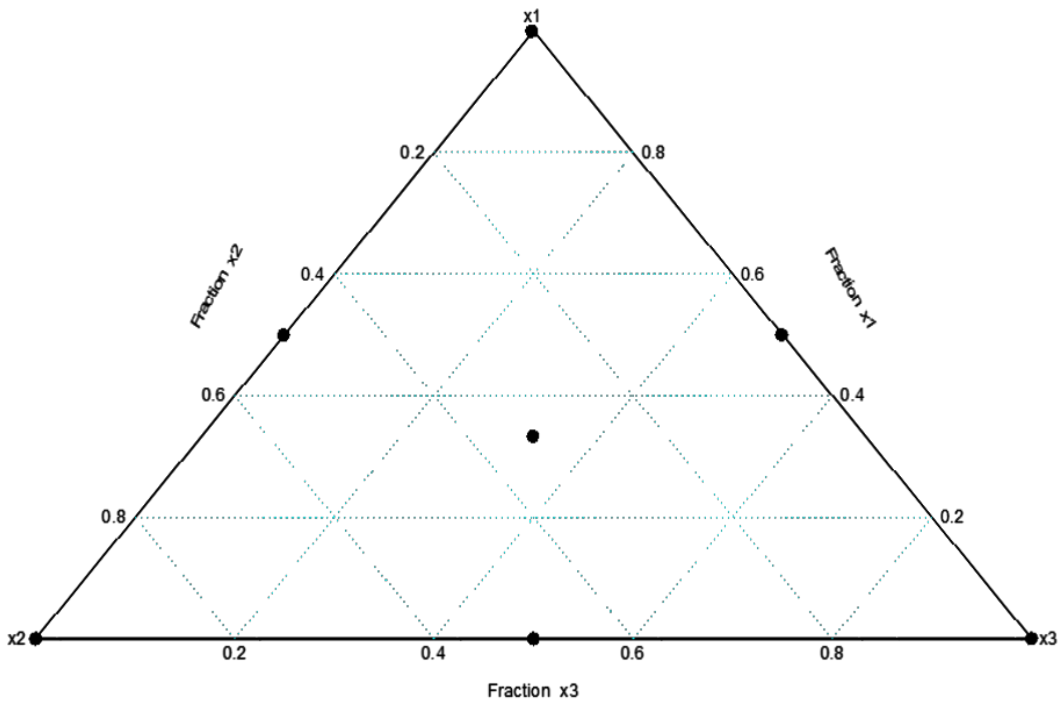

Figure 3.1 Simplex Centroid Design with three components (Source: Lawson and Willden 2016).

Many studies are of individual fuel components rather than complete or partial fuel beds (Trabaud 1976; Scarff and Westoby 2006; Kane *et al.* 2008; Pausas *et al.* 2012). In the study of de Magalhães and Schwilk (2012), controlled flammability tests were conducted on reconstructed monospecific litter beds and on mixed litter beds with litter from three species. Non-additive effects were found to drive flammability of the litter mixture. However, this did not fully explain why mixtures had faster spread rates than individual species and this seemed to be caused by the most flammable species in the mixture (de Magalhães and Schwilk 2012; Della Rocca *et al.* 2018). Other studies have suggested that mineral, silica and volatile contents could be chemical causes of flammability (Mutch and Philpot 1970; Philpot 1970; Alessio *et al.* 2008a, b; Ormeno *et al.* 2009). Since these studies were lacking experimental data to determine chemical causes of flammability, this project has attempted to address these knowledge gaps using litter from typical Sydney Coastal Dry Sclerophyll Forest, a widespread vegetation type found in NSW.

3.2 Experimental design

The SCD was used to determine which forest litter component or mixture was the most flammable. The mixture proportions were determined by using the SCD from the 'mixexp' package (Lawson *et al.* 2016) in the R programming language. This is appropriate for standard mixture designs in unconstrained regions (Cornell 2002; Lawson and Willden 2016). To demonstrate this concept with three components, SCD output using the 'mixexp' package in the R programming language, is shown in Table 3.2.

Table 3.2 Simplex Centroid Design output with three components using the ‘mixexp’ package in R. x1 – twigs, x2 – leaves, x3 – other.

No.	**x1**	**x2**	**x3**
1	1	0	0
2	0	1	0
3	0	0	1
4	0.5	0.5	0
5	0.5	0	0.5
6	0	0.5	0.5
7	$0.\dot{3}$	$0.\dot{3}$	$0.\dot{3}$

A graphical representation of the simplex experimental region can be created with the ‘mixexp’ function DesignPoints (Figure 3.1). To describe this design, if x1 represents twigs, x2 represents leaves and x3 represents other (e.g. bark, fruits), when the proportion of twigs is 100% the other components are zero (Line 1, Table 3.2; the upper vertex of the triangle in Figure 3.1). Equal proportions (50%) of twigs mixed with leaves and no other component represents the mid-point of the left line of the triangle in Figure 3.1 (Line 4, Table 3.2). A mixture of one third twigs, leaves and other is the central point of the triangle in Figure 3.1 (Line 7, Table 3.2).

Litter samples from Halls Creek generally have four fractions including other (hard woody fruit and bark), twigs, leaves and decomposed materials (Table 3.3 and see Chapter 2). Litter samples from Rofe Park had five fractions; leaves, twigs, other, decomposed, cladodes of *Allocasuarina littoralis* (Table 3.4 and see Chapter 2).

Table 3.3 Flammability combinations for litter samples from Halls Creek (H). O – other; T – twigs; L – leaves; Dec or D – decomposed.

Sample	**Other (x1)**	**Twigs (x2)**	**Leaves (x3)**	**Dec (x4)**
HO	1	0	0	0
HT	0	1	0	0
HL	0	0	1	0
HD	0	0	0	1
HOT	0.5	0.5	0	0
HOL	0.5	0	0.5	0
HOD	0.5	0	0	0.5
HTL	0	0.5	0.5	0
HTD	0	0.5	0	0.5
HLD	0	0	0.5	0.5
HOTL	$0.\dot{3}$	$0.\dot{3}$	$0.\dot{3}$	0
HOTD	$0.\dot{3}$	$0.\dot{3}$	0	$0.\dot{3}$
HOLD	$0.\dot{3}$	0	$0.\dot{3}$	$0.\dot{3}$
HTLD	0	$0.\dot{3}$	$0.\dot{3}$	$0.\dot{3}$
HOTLD	0.25	0.25	0.25	0.25

Table 3.4 Flammability combinations for litter samples from Rofe Park (R). Cas or C – *Allocasuarina littoralis*; O – other; T – twigs; L – leaves; Dec or D – decomposed.

Sample	**Cas (x1)**	**Other (x2)**	**Twigs (x3)**	**Leaves (x4)**	**Dec (x5)**
RC	1	0	0	0	0
RO	0	1	0	0	0
RT	0	0	1	0	0
RL	0	0	0	1	0
RD	0	0	0	0	1
RCO	0.5	0.5	0	0	0
RCT	0.5	0	0.5	0	0
RCL	0.5	0	0	0.5	0
RCD	0.5	0	0	0	0.5
ROT	0	0.5	0.5	0	0
ROL	0	0.5	0	0.5	0
ROD	0	0.5	0	0	0.5
RTL	0	0	0.5	0.5	0
RTD	0	0	0.5	0	0.5
RLD	0	0	0	0.5	0.5
RCOT	$0.\dot{3}$	$0.\dot{3}$	$0.\dot{3}$	0	0
RCOL	$0.\dot{3}$	$0.\dot{3}$	0	$0.\dot{3}$	0
RCOD	$0.\dot{3}$	$0.\dot{3}$	0	0	$0.\dot{3}$
RCTL	$0.\dot{3}$	0	$0.\dot{3}$	$0.\dot{3}$	0
RCTD	$0.\dot{3}$	0	$0.\dot{3}$	0	$0.\dot{3}$
RCLD	$0.\dot{3}$	0	0	$0.\dot{3}$	$0.\dot{3}$
ROTL	0	$0.\dot{3}$	$0.\dot{3}$	$0.\dot{3}$	0
ROTD	0	$0.\dot{3}$	$0.\dot{3}$	0	$0.\dot{3}$
ROLD	0	$0.\dot{3}$	0	$0.\dot{3}$	$0.\dot{3}$
RTLD	0	0	$0.\dot{3}$	$0.\dot{3}$	$0.\dot{3}$
RCOTL	0.25	0.25	0.25	0.25	0
RCOTD	0.25	0.25	0.25	0	0.25
RCOLD	0.25	0.25	0	0.25	0.25
RCTLD	0.25	0	0.25	0.25	0.25
ROTLD	0	0.25	0.25	0.25	0.25
RCOTLD	0.2	0.2	0.2	0.2	0.2

The visual representation of SCD with four components is a tetrahedron with four vertices in a three-dimensional space. An experimental design in more than four components is difficult to visualise but is valid mathematically (Leardi 2009). This is also the case with five components. The ‘mixexp’ function DesignPoints does not provide a graphical visual representation for components greater than three (Lawson *et al*. 2016).

3.2.1 Burning/flammability experiments

Litter collected from the Halls Creek and Rofe Park sites was oven-dried at 60°C for 48 h (see Section 2.2.4). This dried litter was then weighed and used to create sample mixtures according to each row of the experimental design described in Tables 3.1 and 3.2. Rofe Park was represented by 31 samples and Halls Creek by 15 samples. Each litter mixture was sealed in airtight containers until burnt.

Approximately 20 g of each mixture was spread evenly over a piece of aluminum foil of known mass used to line a frying pan of 28 cm diameter (Figure 3.2a, b). The litter depth was measured in five different places within the frying pan. A cotton bud drenched with 1 mL of ethanol was placed in the middle of each litter sample and lit with a gas-lighter.

Figure 3.2 Examples of litter mixtures prior to burning: (a) twigs and decomposed material, and (b) leaves and decomposed material, and (c) measuring flame height.

Cotton balls soaked with methylated spirits were used because they are safe, clean and, as they ignite from the top of the fuel bed, are appropriate for simulating ignition from aerial incendiaries and flaming firebrands (Pluckinski and Anderson 2008). This ignition method is also similar to the direct application of a liquid fuel used in the field, such as drip torch ignition for prescribed burns on the urban-bush interface (Pluckinski and Anderson 2008).

A 1 m ruler was placed next to a frying pan for visual flame height (VFH) measurements (Figure 3.2c). Once the litter sample had ignited, stop watches were used to measure time-to-ignition (TTI), duration of visual flaming and burn to completion (BTC). Duration of visual flaming (DVF) was calculated from these metrics. The rate of spread (RS), residual mass fraction (RMF) and volume consumed (VC) were also calculated from these and other measurements (litter depth and size of pan). Each burnt sample was wrapped in foil and weighed to determine the fuel mass after burning.

3.2.2 Statistical analysis

This study was designed as a mixture with four variables for Halls Creek samples and five variables for Rofe Park samples. The SCD determined these mixtures. The blend matrix for Halls Creek samples is shown in Table 3.3 and in Table 3.4 for Rofe Park samples. The statistical model of Brown *et al.* (2015) is used for mixture designs with three components. However, as litter mixtures from Halls Creek had four components and Rofe Park had five components, the statistical equations from the code of Brown *et al.* (2015) were modified to accommodate the required number of components. Furthermore, the code is based on a model matrix that defines the SCD matrix, such that the columns provide the settings for the variables which imposes a limit on the number of variables according to how many columns are in the matrix (Lundstedt *et al.* 1998). A matrix is represented by a collection of single dimension arrays with programming code, so with more mixture components there must be more columns for each variable in the matrix. This meant the number of columns in the arrays had to be increased for the program to have the correct settings for the variables to provide the correct output. Multivariate data analyses and modelling were done using the program of Brown *et al.* (2015) which has the General Blending Model (GBM), the quadratic, cubic and special cubic Scheffé polynomials for mixtures and two of Becker's second-order linear blending models. The statistical model with the best fit for the data was determined. Akaike second-order information criterion for small sample sizes (AIC_c) is a function from the '$AIC_{cmodavg}$' package from the R programming language and statistical environment (Mazerolle 2017). The performance of these statistical models was tested and ranked according to the smallest AIC_c number which was considered to be the best fit for the data. In all cases, the GBM was superior to the other statistical models.

Outputs from the results of the GBM are displayed in the columns of Table 3.7 and Table 3.8. These columns respectively, are 'Metric', 'Coefficient', 'Estimate', 'Std. Error', 'Pr', 'Adj. R^2','*p*' and 'AIC_c' value. The values in the 'Metric' column are the flammability metrics; bulk density (BD), burn to completion (BTC), residual mass fraction (RMF), rate of spread (RS), time to ignition (TTI), volume consumed (VC), visual flame height (VC) and duration of vertical flame (VC). The values in the 'Coefficient' column for Halls Creek are of the litter components; x1 other, x2 twigs, x3 leaves and x4 decomposed. The values for Rofe Park are; x1 *Allocasuarina littoralis*, x2 other, x3 twigs, x4 leaves and x5 decomposed. The GBM is a statistical model that will maximise the fit by changing the value of the exponents of the equations in the 'Coefficients' column. The individual coefficient or equation with coefficients for each row depict the equation for the fit of the response surface. These equations also demonstrate the coefficient or interactions between coefficients that either increase or decrease flammability depending on whether the estimate value is positive or negative. The positive values in the 'Estimate' column demonstrate the coefficient or interaction between coefficients enhance flammability. Negative values indicate a decreasing effect on flammability. The 'Pr' values depict the statistical significance of the flammability effect by the coefficients. The significance codes are: 0 '***', 0.001 '**', 0.001 '*', 0.05 ' ', 0.1 ' ', 1. So only Pr values < 0.05 are statistically significant. Only significant values are shown in Table 3.7 and Table 3.8, respectively. 'Std. Error' is the standard error or uncertainty. 'Adj. R^2' is the squared bias. A high R^2 value, e.g. $R^2 = 1$, means there are little or no residuals in the model when fitted against the dataset. For a model to achieve parsimony there is a trade-off between the uncertainty and squared bias values. The AIC_c value takes these variables into account to find the model that is the best fit for the data. Only the models with the smallest AIC_c values are shown in Table 3.7 and Table 3.8, respectively. In the *p* column are *p*-values that summarise the evidence against the null hypothesis that there is no relationship between the metric and the predictor variables.

A non-additive effect is when a component dominates flammability to a greater extent than the proportion of its weight in the mixture (van Altena *et al.* 2012). Flammability metric for samples from Halls Creek are shown in Table 3.7 and in Table 3.8 for samples from Rofe Park. For each flammability metric there are individual coefficients and, in another row, an equation with more than one coefficient. If an individual coefficient has a statistically significant positive estimate value and this is also the case for a non-linear equation with this individual coefficient in the mixture, then this coefficient is producing a non-additive flammability effect on the mixture.

3.2.3 Model optimisation

Optimisations were run using the R program ‘nloptr’ (Johnson, The NLopt non- linear optimisation package, http://ab-initio.mit.edu/nlopt) to determine the fit of the data within the response surface. Response surface methodology is the only way to determine the optimum value for mixture design experiments (Lunstedt *et al*. 1998). The mixture design model has quadratic terms so non-linear relationships between the experimental variables and responses could be determined (Lundstedt *et al.* 1998). Therefore, non-linear optimisation methods are required to solve the maxima and minima in the design model. If there was a perfect fit of the statistical model of Brown *et al*. (2015), the maxima and minima of the model fits would equal the measured maxima and minima of the measurements. Therefore, the proportions of the different fuel fractions that produced the measured maxima and minima were used as the starting point for the algorithm to determine whether the minimum or maximum optimum could be solved for the response surface. If the algorithm suggested other feasible solutions then they were also tested to determine whether it was a feasible solution. ‘nloptr’ interfaces to the NLopt library for non-linear optimisation, which enables optimisation routines and algorithms to be used (Ypma *et al*. 2018). If the global solver, ‘Augmented Lagrangian Algorithm’ (auglag), and the local solver, ‘Low-storage version of the Broyden-Fletcher-Goldfarb-Shanno’ (BFGS), method did not produce a feasible solution then the global solver (NLOPT_LN_AUGLAG) and local solver (Constrained Optimisation by Linear Approximations; COBYLA) were used instead. Refer to the Appendix for definitions. A failure was identified by ‘NLOPT_FAILURE: Generic failure code’ (Ypma *et al*. 2018). In contrast, successful outputs were considered when

'NLOPT_MAXEVAL_REACHED: Optimisation stopped' because maxeval (above) was reached', 'NLOPT_XTOL_REACHED: Optimisation stopped because xtol_rel or xtol_abs (above) was reached', 'NLOPT_FTOL_REACHED: Optimisation stopped because ftol_rel or ftol_abs (above) was reached' or 'NLOPT_ROUNDOFF_LIMITED: Roundoff errors led to a breakdown of the optimisation algorithm. In this case, the returned minimum may still be useful (e.g. this error occurs in NEWUOA if one tries to achieve a tolerance too close to machine precision) (Ypma *et al*. 2018).

3.3 Results

3.3.1 Measures of flammability

Flammability and relevant descriptive variables of each of the litter mixtures are presented in Table 3.5 for Halls Creek and in Table 3.6 for Rofe Park. Bulk density (BD) was the greatest for the decomposed fraction from Rofe Park (RD) at 95 kg m^{-3} (Table 3.6). The next greatest BD value was for twigs from Halls Creek (HT) at 77 kg m^{-3} (Table 3.5). This is consistent with a large amount of coarse woody debris that was found at the site (see Chapter 2, Table 2.1). The lowest BD for both sites was leaves from Halls Creek (HL) at 15 kg m^{-3} (Table 3.5) and *Allocasuarina littoralis* cladodes from Rofe Park (RC) (Table 3.6).

Time to ignition (TTI) was rapid for litter from both sites that contained leaves (Tables 3.5 and 3.6). The decomposed fraction from Halls Creek (HD) and other (RO) and decomposed fractions (ROD) from Rofe Park did not ignite or did not burn sufficiently to affect the residual mass fraction (RMF). The time for burn to completion (BTC) and volume consumed (VC) values were usually correlated. When litter burns to completion a greater volume of fuel is consumed. Leaves caused the most rapid BTC and VC times for both sites. For 'Casuarina' litter from Rofe Park (RC), the value of BTC was 64 s (Table 3.6) and

1.33×10^{-3} m^3 for VC (Table 3.6). The fastest time for BTC (91 s) and VC (0) for Halls Creek was for a mixture of other, twigs and leaves (HOTL) (Table 3.5). The slowest time for BTC and VC for both sites were recorded for mixtures of leaves and twigs. For litter from Rofe Park, 'Casuarina', other and twigs (RCOT) had values of 234 s for BTC and 1.56×10^{-3} m^3 for VC (Table 3.6) and for Halls Creek, twigs and leaves (HTL) required 335 s for BTC and had 8.00×10^{-4} m^3 for VC (Table 3.5). There were samples from both Halls Creek (60 %) and Rofe Park (35 %) that ignited but rapidly went out so there were zero values for RS and VC and 100% for RMF (Table 3.5 and Table 3.6). Halls Creek had the greatest proportion of these samples.

Rate of spread (RS) increased when *Allocasuarina littoralis* was included in litter mixtures as indicated by rates that were twice as fast for Rofe Park compared to Hall Creek (Tables 3.5 and 3.6).

Many of the litter combinations from Halls Creek did not burn whereas most of them from Rofe Park did burn. Residual mass fraction (RMF) was therefore 100% for several litter combinations from both sites (Tables 3.5 and 3.6). Residual mass fractions of litter combinations from Halls Creek that did burn included other and twigs (HOT, 0.14%) and other (HO, 21%) (Table 3.5). Litter combinations from Rofe Park with the lowest RMF had twigs in common and included twigs (RT, 3.5%), twigs and leaves (RTL, 6.51%) and other, twigs and leaves (ROTL, 9%) (Table 3.6).

Table 3.5 Flammability metrics for combinations of litter components collected from Halls Creek (H) as specified by a simplex centroid design. The materials include: O – other ; T – twigs; L – leaves; Dec or D – decomposed. Flammability metrics are bulk density BD – bulk density; TTI – time to ignition; VFH – visual flame height; RS – rate of spread; VC – volume consumed; RMF – residual mass fraction; BTC – burn to completion; DVF – duration of vertical flame.

Sample	**BD ($kg\ m^{-3}$)**	**TTI (s)**	**VFH (m)**	**RS ($m\ s^{-1}$)**	**VC (m^3)**	**RMF (%)**	**BTC (s)**	**DVF (s)**
HO	28	2	0.22	2.50×10^{-3}	2.62×10^{-4}	21.38	202	200
HT	77	28	0.02	0	0	100	120	92
HL	15	7	0.30	1.80×10^{-3}	1.13×10^{-3}	62.88	179	172
HD	53	0	0	0	0	100	0	0
HOT	31	6	0.16	4.10×10^{-3}	7.14×10^{-4}	0.14	97	91
HOL	21	1	0.24	1.90×10^{-3}	9.73×10^{-4}	38.71	223	222
HOD	36	3	0.01	0	0	100	5	2
HTL	26	7	0.20	1.50×10^{-3}	8.00×10^{-4}	45.41	335	328
HTD	74	24	0.01	0	0	100	171	147
HLD	37	5	0.12	0	0	100	110	105
HOTL	27	7	0.08	0	0	100	91	112
HOTD	40	9	0.03	0	0	100	12	3
HOLD	28	1	0.01	0	0	100	0	0
HTLD	45	15	0	0	0	100	0	0
HOTLD	24	1	0.11	2.50×10^{-3}	0	100	119	90

Table 3.6 Flammability metrics for combinations of materials collected from Rofe Park (R) as specified by a simplex centroid design. The materials include: C – *Allocasuarina littoralis*; O – other; T – twigs; L – leaves; D – decomposed. and decomposed materials (D). Flammability metrics are bulk density BD – bulk density; TTI – time to ignition; VFH – visual flame height; RS – rate of spread; VC – volume consumed; RMF – residual mass fraction; BTC – burn to completion; DVF – duration of vertical flame. Flammability values are rounded to whole numbers for BD, TTI, BTC and VC.

Sample	**BD ($kg\ m^{-3}$)**	**TTI (s)**	**VFH (m)**	**RS ($m\ s^{-1}$)**	**VC (m^3)**	**RMF (%)**	**BTC (s)**	**DVF (s)**
RC	15	3	0.37	6.10×10^{-3}	1.33×10^{-3}	60.89	64	61
RO	28	3	0.03	0	7.14×10^{-4}	33.77	115	112
RT	29	35	0.10	9.00×10^{-4}	6.77×10^{-4}	3.52	193	158
RL	21	3	0.15	1.90×10^{-3}	9.73×10^{-4}	50.2	170	167
RD	95	25	0.05	0	0	100	0	0
RCO	11	1	0.27	5.20×10^{-3}	1.86×10^{-3}	46.35	85	84
RCT	11	2	0.33	6.70×10^{-3}	1.85×10^{-3}	61	147	145
RCL	17	2	0.30	4.50×10^{-3}	1.18×10^{-3}	56.89	113	111
RCD	13	1	0.31	4.20×10^{-3}	1.54×10^{-3}	26.3	90	89
ROT	22	3	0.14	0	0	100	149	146
ROL	24	2	0.16	0	0	100	146	144
ROD	46	0	0	0	0	100	0	0
RTL	24	3	0.18	2.20×10^{-3}	8.37×10^{-4}	6.51	184	181
RTD	54	12	0.15	0	0	100	0	0
RLD	32	11	0.15	0	0	100	120	109
RCOT	12	1	0.24	3.40×10^{-3}	1.56×10^{-3}	58.71	234	233
RCOL	11	1	0.38	4.50×10^{-3}	1.56×10^{-3}	85.34	206	205
RCOD	14	1.5	0.29	5.90×10^{-3}	1.30×10^{-3}	23.09	114	112.5
RCTL	8	2	0.43	4.70×10^{-3}	2.02×10^{-3}	47.35	207	205
RCTD	19	2	0.18	2.30×10^{-3}	9.36×10^{-4}	21.8	159	157
RCLD	10	1	0.41	5.00×10^{-3}	1.71×10^{-3}	42.67	140	139
ROTL	19	3	0.20	2.00×10^{-3}	9.61×10^{-4}	9.09	137	134
ROTD	27	40	0.12	0	0	100	0	0
ROLD	28	8	0.11	0	0	100	87	79
RTLD	34	12	0.06	0	0	100	92	80
RCOTL	17	5	0.23	4.70×10^{-3}	1.22×10^{-3}	28.83	135	130
RCOTD	28	3	0.12	1.80×10^{-3}	0	100	134	131
RCOLD	22	4	0.28	4.70×10^{-3}	9.24×10^{-4}	21.91	85	81
RCTLD	15	6	0.26	7.40×10^{-3}	1.40×10^{-3}	27.13	104	98
ROTLD	49	8	0.04	0	0	100	30	22
RCOTLD	19	3	0.23	3.70×10^{-3}	1.13×10^{-3}	10.58	95	92

3.3.2 Statistical modelling

Using the AIC_c criterion, the model of Brown *et al.* (2015) provided the best fit for all of the data.

Halls Creek

The bulk density (BD) model was a good fit against the experimental data (adjusted $R^2 = 9.89\times10^{-1}$, $p = 2.78\times10^{-9}$). The equation consisted of a mixture of exponents that are linear and to the power of 0.5. There was a particularly strong negative non-additive effect between other and twigs as litter components (estimate = -5.48×10^{1}). There was another very strong non-additive effect between twigs and leaves (estimate = -1.36×10^{3}). This equation consisted of a mixture of constant and cubic exponents. All variables provided a significant positive contribution to BD. Twigs contributed the most to BD and leaves contributed the least (Table 3.7).

The burn to completion (BTC) model was a good fit against the experimental data (adjusted $R^2 = 8.08\times10^{-1}$, $p = 3.20\times10^{-3}$) and there were no non-additive effects. Other (estimate = 1.35×10^{2}) and leaves (estimate = 1.93×10^{2}) as litter components provided a significant positive contribution to BTC (Table 3.7).

The residual mass fraction (RMF) model was a good fit against the experimental data (adjusted $R^2 = 9.48\times10^{-1}$, $p = 2.97\times10^{-6}$). There was a very strong negative non-additive effect between litter components of other and twigs (estimate = -8.70×10^{3}) and the equation consisted of constant and cubic exponents. There was also a strong positive non-additive effect between other and twigs as litter components (estimate = 1.70×10^{3}), with the relevant equation consisting of a mixture of exponents that were cubic and to the power of 1.5 (Table 3.7).

The rate of spread (RS) model was a perfect fit of the experimental data (adjusted $R^2 = 1$, $p = 9.51\times10^{-4}$). Twigs had the strongest positive effect on rate of spread (estimate = 5.70×10^{-1}) and the next strongest positive effect was from other (estimate = 2.50×10^{-1}). There were non-additive effects with very significant negative interactions between twigs and leaves (estimate = -1.02×10^{1}). The equation consisted of a mixture of exponents that were cubic and to the power of 2.5. There was another strong negative interaction between other and leaves (estimate = -1.60×10^{0}) and the equation consisted a mixture of constant and cubic exponents (Table 3.7).

The time to ignition (TTI) model was an adequate fit against the experimental data (adjusted $R^2 = 0.871$, $p = 1.28\times10^{-4}$) and there were no non-additive effects. Twigs had the strongest positive effect (estimate = 2.42×10^{1}) and the decomposed fraction also had a strong positive effect (estimate = 1.60×10^{1}) (Table 3.7). The volume consumed (VC) model was a particularly good fit against the experimental data (adjusted $R^2 = 0.929$, $p = 1.78\times10^{-1}$), however, there were no significant effects (Table 3.7). The model for visual flame height (VFH) was a good fit against the experimental data (adjusted $R^2 = 0.903$, $p = 1.07\times10^{-4}$) and there were no non-additive effects. Leaves (estimate = 3.18×10^{-1}) and other (estimate = 2.11×10^{-1}) provided strong positive effects (Table 3.9). The duration of vertical flame (DVF) model was an adequate fit against the experimental data (adjusted $R^2 = 0.806$, $p = 3.30\times10^{-3}$) and there were no non-additive effects. Leaves had the strongest positive effect (estimate = 1.88×10^{2}) (Table 3.7).

The volume consumed (VC) model was an excellent fit against the experimental data (adjusted $R^2 = 0.929$, $p = 1.78\times10^{-1}$). There was no non-additive effect or significant values (Table 3.7).

The visual flame height (VFH) model was a good fit against the experimental data (adjusted $R^2 = 0.903$, $p = 1.07\times10^{-4}$). There was no non-additive effect. Leaves had the strongest positive effect (estimate = 3.18×10^{-1}) and other also had a strong positive effect (estimate = 2.11×10^{-1}) (Table 3.7).

The duration of vertical flame (DVF) model was an adequate fit against the experimental data (adjusted $R^2 = 0.806$, $p = 3.30\times10^{-3}$). There was no non-additive effect. Leaves had the strongest positive effect (estimate = 1.88×10^{2}) and other also had a strong positive effect (estimate = 1.38×10^{2}) (Table 3.7).

Table 3.7 Summary table of model outputs for Halls Creek for BD – bulk density; BTC – burn to completion; RMF – residual mass fraction; RS – rate of spread; TTI – time to ignition; VC – volume consumed; VFH – visual flame height; DVF – duration of vertical flame. x1 – other; x2 – twigs; x3 – leaves, x4 is decomposed; Std. Error – standard error; Pr – Pr(>|t|); Adj. R^2 – Adjusted R-squared; p – p-value; AIC_c – Akaike information criterion for small sample sizes. Statistical significance codes: 0 – ***; 0.001 – **'; 0.01 – *,

Metric	Coefficient	Estimate	Std. Error	Pr	Adj. R^2	*p*	AIC_c value
BD	x1	2.55×10^{1}	3.53×10^{0}	4.97×10^{-5} ***	9.89×10^{-1}	2.78×10^{-9}	111
	x2	7.97×10^{1}	3.83×10^{0}	6.34×10^{-9} ***			
	x3	1.49×10^{1}	3.41×10^{0}	1.79×10^{-3} **			
	x4	5.38×10^{1}	3.20×10^{0}	4.25×10^{-8} ***			
	$I(x1^{1} \times x2^{0.5}/(x1 + x2 + 0.001)^{1})$	-5.48×10^{1}	9.52×10^{0}	2.73×10^{-4} ***			
	$I(x2^{3} \times x3^{3}/(x2 + x3 + 0.001)^{0})$	-1.36×10^{3}	3.26×10^{2}	2.44×10^{-3} **			
BTC	x1	1.35×10^{2}	5.47×10^{1}	4.27×10^{-2} *	8.08×10^{-1}	3.20×10^{-3}	159
	x3	1.93×10^{2}	5.87×10^{1}	1.35×10^{-2} *			
RMF	x2	8.67×10^{1}	1.59×10^{1}	4.03×10^{-4} ***	9.48×10^{-1}	2.97×10^{-6}	154
	x3	6.26×10^{1}	1.44×10^{1}	1.82×10^{-3} **			
	x4	1.22×10^{2}	1.44×10^{1}	1.39×10^{-5} ***			
	$I(x1^{3} \times x2^{3}/(x1 + x2 + 0.001)^{0})$	-8.70×10^{3}	2.30×10^{3}	4.30×10^{-3} **			
	$I(x1^{3} \times x2^{1.5}/(x1 + x2 + 0.001)^{3})$	1.70×10^{3}	6.96×10^{2}	3.75×10^{-2} *			

Table 3.7 continued.

Metric	Coefficient	Estimate	Std. Error	Pr	Adj. R^2	*p*	AICc value
RS	x1	2.50×10^{-1}	3.45×10^{-4}	8.77×10^{-4} ***	1	9.51×10^{-4}	-161
	x2	5.70×10^{-1}	7.63×10^{-4}	8.51×10^{-4} ***			
	x3	1.80×10^{-1}	3.46×10^{-4}	1.22×10^{-3} **			
	$I(x2^3 \times x3^{2.5}/(x2 + x3 + 0.001)^3)$	-1.02×10^{1}	2.49×10^{-2}	1.55×10^{-3} **			
	$I(x1^3 \times x3^3/(x1 + x3 + 0.001)^0)$	-1.60×10^{0}	2.72×10^{-2}	1.08×10^{-2} *			
TTI	x2	2.42×10^{1}	3.09×10^{0}	2.65×10^{-5} ***	8.71×10^{-1}	1.28×10^{-4}	98
	x4	1.60×10^{1}	5.07×10^{0}	1.16×10^{-2} *			
VC	x4 not defined because of singularities		No significant values		9.29×10^{-1}	1.78×10^{-1}	-128
VFH	x1	2.11×10^{-1}	3.84×10^{-2}	5.78×10^{-4} ***	9.03×10^{-1}	1.07×10^{-4}	-23
	x3	3.18×10^{-1}	3.88×10^{-2}	3.63×10^{-5} ***			
DVF	x1	1.38×10^{2}	5.27×10^{1}	3.41×10^{-2} *	8.06×10^{-1}	3.30×10^{-3}	158
	x3	1.88×10^{2}	5.66×10^{1}	1.26×10^{-2} *			

Rofe Park

The BD model was a particularly good fit against the experimental data (adjusted R^2 = 0.9563, $p = 2.79\times10^{-15}$). There were non-additive effects with the decomposed fraction being strongly influential (estimate = 9.24×10^{1}). Negative interactions were identified between the decomposed fraction and *Allocasuarina littoralis,* leaves and other fractions with the strongest negative interaction with *Allocasuarina littoralis* (estimate = -1.04×10^{2}) (Table 3.8).

The BTC model was an excellent fit against the experimental data (adjusted R^2 = 0.9733, $p = 6.91\times10^{-14}$) but there were non-additive effects. The strongest positive effect of individual components was from twigs (estimate = 2.03×10^{2}). *Allocasuarina littoralis* cladodes had a very strong positive effect and there was a particularly significant interaction between the litter components of *Allocasuarina littoralis,* other and twigs (estimate = 4.93×10^{5}). There was a particularly strong non-additive effect between leaves and decomposed (estimate = -1.69×10^{2}). The equation consisted of exponents that were cubic and to the power of 0.5 and 1.5 (Table 3.8).

The residual mass fraction RMF model was a good fit against the experimental data (adjusted R^2 = 0.9157, $p = 3.27\times10^{-7}$). There was a non-additive effect with *Allocasuarina littoralis* (estimate = 6.82×10^{1}) and this litter component, along with leaves, had the strongest positive effect. In contrast, there was a negative interaction between twigs and decomposed fractions (estimate = -5.21×10^{1}) with the equation consisting of a mixture of exponents that were constant and to the power of 0.5 (Table 3.8).

The RS model was an excellent fit against the experimental data (adjusted $R^2 = 0.9668$, $p = 2.82\times10^{-8}$). There was a non-additive effect with *Allocasuarina littoralis* having the strongest positive influence (estimate = 5.79×10^{-1}). The model of RS identified three of the components, *Allocasuarina littoralis*, other and leaves as having strong positive effects and positive interactions were identified between *Allocasuarina littoralis* and leaves, *Allocasuarina littoralis* and twigs, and other and decomposed. The interaction between *Allocasuarina littoralis* and leaves was found to be particularly strongly positive (estimate = 6.13×10^{-1}) and consisted of a mixture of exponents that were cubic and to the powers of 0.5 and 1.5. In contrast, there was a negative interaction between *Allocasuarina littoralis* and other (estimate = - 1.07×10^{-1}) and the equation consisted of a mixture of exponents that were cubic and to the power of 0.5 (Table 3.8).

The TTI model was an adequate fit against the experimental data (adjusted $R^2 = 0.8329$, $p = 5.05\times10^{-7}$). There was a non-additive effect with twigs having the strongest influence (estimate = 3.66×10^{1}). The model of TTI identified two of the components, twigs and decomposed, as having a strong positive effect (estimate = -1.10×10^{3}). The equation describing this interaction was a mixture of constant and cubic components. In contrast, there were very strong negative interactions between *Allocasuarina littoralis* and the decomposed fraction (estimate = -2.79×10^{1}) and the equation consisted of a mixture of exponents that were constant and to the power of 0.5 (Table 3.8).

The VC model was a particularly good fit against the experimental data (adjusted $R^2 = 0.967$, $p = 4.80\times10^{-9}$). There was a non-additive effect with *Allocasuarina littoralis* having the strongest influence (estimate = 1.42×10^{-3}) and the decomposed fraction was also influential (estimate = 1.56×10^{-3}). The model of VC identified all five litter components as having a strong positive effect. There were very strong positive interactions between *Allocasuarina littoralis* and twigs (estimate = 3.91×10^{-3}) and the equation consisted of exponents that were constant and to the power of 0.5 and 1.5. In contrast, the interaction between twigs and decomposed was negative (estimate = -5.24×10^{-1}) and consisted of constant and cubic exponents (Table 3.8).

The VFH model was a particularly good fit against the experimental data (adjusted R^2 = 0.9624, $p = 1.50\times10^{-14}$). There was a non-additive effect with *Allocasuarina littoralis* having the strongest influence (estimate = 4.31×10^{-1}). The model of VFH also identified all five litter components as having a strong positive effect. A positive interaction was identified between *Allocasuarina littoralis* and leaves (estimate = 2.71×10^{2}) and the equation consisted of a mixture of exponents that were constant, cubic and to the power of 2.5 (Table 3.8).

The DVF model was an excellent fit against the experimental data (adjusted R^2 = 0.9723, $p = 6.44\times10^{-13}$). There was a non-additive effect with *Allocasuarina littoralis* having the strongest influence (estimate = 7.14×10^{1}). The model of DVF identified all five litter components as having a strong positive effect.

The DVF model had a positive interaction that was identified between *Allocasuarina littoralis*, other and twigs, and *Allocasuarina littoralis* and leaves. In contrast, negative interactions were identified between leaves and decomposed, *Allocasuarina littoralis*, other and twigs, and *Allocasuarina littoralis* and decomposed. The interaction between leaves and decomposed was particularly strongly negative (estimate = -2.24×10^2) and consisted of a mixture of exponents that were cubic and to the power of 0.5 and 1.5 (Table 3.8).

Another way to demonstrate non-additive effects for VFH are shown in Table 3.11. Visual flame height values were compared for Halls Creek (H) and Rofe Park (R) samples without and with *Allocasuarina littoralis* (C). The percentage differences are variable in the first comparison (HR) but have consistently positive values when *Allocasuarina littoralis* was added into Rofe Park litter samples. However, the size of that positive increase is not consistent alluding to non-additive effects (Table 3.11).

Table 3.8 Summary table of model outputs for Rofe Park for bulk density BD-bulk density; TTI – time to ignition; VFH – visual flame height; RS – rate of spread; VC – volume consumed; RMF – residual mass fraction; BTC – burn to completion; DVF – duration of vertical flame. x1 – *Allocasuarina littoralis*; x2 – other; x3 – twigs; x4 – leaves; x5 – decomposed; Std. Error – standard error; Pr – Pr(>|t|); Adj. R^2 – Adjusted R-squared; p – p-value; AIC_c – Akaike information criterion for small sample sizes. Statistical significance codes: 0 – ***; 0.001 – **; 0.01 –*.

Metric	Coefficient	Estimate	Std. Error	Pr	Adj. R^2	*p*	AIC_c value
BD	x2	2.34×10^{1}	4.51×10^{0}	2.89×10^{-5} ***	0.9563	2.79×10^{-15}	220
	x3	2.09×10^{1}	4.08×10^{0}	3.35×10^{-5} ***			
	x4	2.05×10^{1}	4.46×10^{0}	1.26×10^{-4} ***			
	x5	9.24×10^{1}	5.59×10^{0}	2.96×10^{-14} ***			
	$I(x1^{1} \times x5^{0.5}/(x1 + x5 + 0.001)^{0})$	-1.04×10^{2}	1.57×10^{1}	9.01×10^{-7} ***			
	$I(x4^{2.5} \times x5^{0.5}/(x4 + x5 + 0.001)^{0})$	-2.04×10^{2}	5.56×10^{1}	1.28×10^{-3} **			
	$I(x2^{3} \times x5^{2.5}/(x2 + x5 + 0.001)^{3})$	-6.49×10^{2}	3.02×10^{2}	4.27×10^{-2} *			
BTC	x1	6.24×10^{1}	1.80×10^{1}	2.74×10^{-3} **	0.9733	6.91×10^{-14}	268
	x2	1.24×10^{2}	1.67×10^{1}	6.71×10^{-7} ***			
	x3	2.03×10^{2}	1.67×10^{1}	3.97×10^{-10} ***			
	x4	1.61×10^{2}	1.64×10^{1}	1.13×10^{-8} ***			
	x5	1.56×10^{2}	3.08×10^{1}	8.03×10^{-5} ***			
	$I(x4^{1.5} \times x5^{0.5}/(x4 + x5 + 0.001)^{3})$	-1.69×10^{2}	3.20×10^{1}	5.00×10^{-5} ***			
	$I(x1^{2.5} \times x2^{2.5} \times x3^{2.5})$	4.93×10^{5}	9.88×10^{4}	9.63×10^{-5} ***			
	$I(x1^{2}\times x4^{0.5}/(x1 + x4 + 0.001)^{3})$	2.33×10^{2}	5.40×10^{1}	4.19×10^{-4} ***			
	$I(x2^{2} \times x3^{0.5}/(x2 + x3 + 0.001)^{3})$	-1.32×10^{2}	5.16×10^{2}	1.94×10^{-2} *			

Table 3.8 continued.

Metric	Coefficient	Estimate	Std. Error	Pr	Adj. R^2	***p***	AIC_c value
RMF	x1	6.82×10^{1}	$9.94\times10^{0.}$	1.16×10^{-5} ***	0.9157	3.27×10^{-7}	177
	x2	3.54×10^{1}	9.57×10^{0}	2.65×10^{-3} **			
	x4	5.79×10^{1}	1.03×10^{1}	8.25×10^{-5} ***			
	$I(x3^{0.5} \times x4^{0.5}/(x3 + x4 + 0.001)^{0})$	-5.21×10^{1}	2.09×10^{1}	2.72×10^{-2} *			
	$I(x1^{1.5} \times x3^{0.5}/(x1 + x3 + 0.001)^{0})$	1.18×10^{2}	5.00×10^{1}	3.46×10^{-2} *			
RS	x1	5.79×10^{-1}	6.97×10^{-2}	4.58×10^{-6} ***	0.9668	2.82×10^{-8}	-11
	x2	4.75×10^{-1}	1.43×10^{-1}	6.65×10^{-3} **			
	x4	1.60×10^{-1}	6.90×10^{-2}	4.11×10^{-2} *			
	$I(x1^{1.5} \times x4^{0.5}/(x1 + x4 + 0.001)^{3})$	6.13×10^{-1}	1.12×10^{-1}	1.88×10^{-4} ***			
	$I(x1^{3} \times x3^{3}/(x1 + x3 + 0.001)^{0})$	1.99×10^{1}	6.14×10^{0}	7.77×10^{-3} **			
	$I(x1^{0.5} \times x2^{0.5}/(x1 + x2 + 0.001)^{3})$	-1.07×10^{-1}	3.12×10^{-2}	5.82×10^{-3} **			
	$I(x2^{3} \times x5^{3}/(x2 + x5 + 0.001)^{0})$	2.06×10^{2}	7.36×10^{1}	1.72×10^{-2} *			

Table 3.8 continued.

Metric	Coefficient	Estimate	Std. Error	Pr	Adj. R^2	p	AIC_c value
TTI	x3	3.66×10^{1}	4.57×10^{0}	1.66×10^{-7} ***	0.8329	5.05×10^{-7}	208
	x5	2.40×10^{1}	4.15×10^{0}	1.46×10^{-5} ***			
	$I(x1^{0.5} \times x5^{0.5}/(x1 + x5 + 0.001)^{0})$	-2.79×10^{1}	7.35×10^{0}	1.22×10^{-3} **			
	$I(x2^{3} \times x5^{3}/(x2 + x5 + 0.001)^{0})$	5.59×10^{3}	2.36×10^{3}	2.87×10^{-2} *			
	$I(x1^{3} \times x3^{2}/(x1 + x3 + 0.001)^{3})$	-5.72×10^{2}	1.57×10^{2}	1.73×10^{-3} **			
	$I(x3^{2.5} \times x4^{0.5}/(x3 + x4 + 0.001)^{0})$	-1.35×10^{2}	4.34×10^{1}	5.93×10^{-3} **			
	$I(x3^{3} \times x5^{3}/(x3 + x5 + 0.001)^{0})$	-1.10×10^{3}	3.67×10^{2}	7.21×10^{-3} **			
	$I(x2^{3} \times x3^{3}/(x2 + x3 + 0.001)^{0})$	-9.45×10^{2}	3.62×10^{2}	1.72×10^{-2} *			
VC	x1	1.42×10^{-3}	2.17×10^{-4}	2.74×10^{-5} ***	0.967	4.80×10^{-9}	-250
	x2	6.75×10^{-4}	2.18×10^{-4}	9.14×10^{-3} **			
	x3	7.04×10^{-4}	2.14×10^{-4}	6.40×10^{-3} **			
	x4	1.16×10^{-3}	1.83×10^{-4}	3.77×10^{-5} ***			
	x5	1.56×10^{-3}	3.68×10^{-4}	1.14×10^{-3} **			
	$I(x1^{1.5} \times x3^{0.5}/(x1 + x3 + 0.001)^{0})$	3.91×10^{-3}	1.05×10^{-3}	2.89×10^{-3} **			
	$I(x1^{3} \times x2^{2.5}/(x1 + x2 + 0.001)^{2})$	3.68×10^{-2}	1.30×10^{-2}	1.55×10^{-2} *			
	$I(x3^{3} \times x5^{3}/(x3 + x5 + 0.001)^{0})$	-5.24×10^{-1}	2.10×10^{-1}	2.81×10^{-2} *			

Table 3.8 continued.

Metric	Coefficient	Estimate	Std. Error	Pr	Adj. R^2	p	AIC_c value
VFH	x1	4.31×10^{-1}	3.32×10^{-2}	1.71×10^{-11} ***	0.9624	1.50×10^{-14}	-79
	x2	8.99×10^{-2}	3.07×10^{-2}	8.05×10^{-3} **			
	x3	1.54×10^{-1}	3.10×10^{-2}	6.76×10^{-5} ***			
	x4	1.69×10^{-1}	3.32×10^{-2}	4.78×10^{-5} ***			
	x5	1.05×10^{-1}	3.27×10^{-2}	4.10×10^{-3} **			
	$I(x1^3 \times x4^3/(x1 + x4 + 0.001)^0)$	-3.87×10^{2}	1.60×10^{2}	2.44×10^{-2} *			
	$I(x1^3 \times x4^{2.5}/(x1 + x4 + 0.001)^0)$	2.71×10^{2}	1.14×10^{2}	2.70×10^{-2} *			
DVF	x1	7.14×10^{1}	1.83×10^{1}	1.12×10^{-3} **	0.9723	6.44×10^{-13}	272
	x2	1.12×10^{2}	1.57×10^{1}	1.78×10^{-6} ***			
	x3	1.72×10^{2}	1.57×10^{1}	4.12×10^{-9} ***			
	x4	1.68×10^{2}	1.70×10^{1}	1.79×10^{-8} ***			
	x5	1.68×10^{2}	3.13×10^{1}	5.36×10^{-5} ***			
	$I(x4^{1.5} \times x5^{0.5}/(x4 + x5 + 0.001)^3)$	-2.24×10^{2}	3.29×10^{1}	3.12×10^{-6} ***			
	$I(x1^{2.5} \times x2^{2.5} \times x3^{2.5})$	1.06×10^{7}	3.82×10^{6}	1.28×10^{-2} *			
	$I(x1^2 \times x4^{0.5}/(x1 + x4 + 0.001)^3)$	3.12×10^{2}	5.95×10^{1}	6.59×10^{-5} ***			
	$I(x1^{2.5} \times x2^{2.5} \times x3^2)$	-5.89×10^{6}	2.20×10^{6}	1.59×10^{-2} *			
	$I(x1^3 \times x4^3/(x1 + x4 + 0.001)^0)$	-3.83×10^{3}	1.72×10^{3}	3.95×10^{-2} *			

3.3.3 Optimisation results

The successful optimisation results demonstrated a good fit of the data within the response surface (Tables 3.9 and 3.10). There was more than one optimal solution with the SCD as the domain is a regular figure with many vertices as components and a space with dimensionality equal to the number of components minus one (Leardi 2009). This means the SCD for Halls Creek with four litter components was represented with a response surface of four vertices in a three-dimensional space. The SCD for Rofe Park was represented with a response surface of five vertices in a four-dimensional space. Since none of the flammability metrics can be less than zero there can be more than one optimum within these complex response surfaces. The global and local solvers found the optimum values were the maxima and minima from the data from Halls Creek and Rofe Park and included all the flammability metrics: BD, BTC, RMF, RS, TTI, DVF, VC and VFH.

Table 3.9 Optimisation of the ideal mixture for (a) maximum and (b) minimum flammability for litter from Halls Creek for BD – bulk density; BTC – burn to completion; RMF – residual mass fraction; RS – rate of spread; TTI – time to ignition; VC – volume consumed; VFH – visual flame height; DVF – duration of vertical flame; x1 – other; x2 – twigs; x3 – leaves, x4 – decomposed.

(a)

Metric	x1	x2	x3	x4
BD	0	1	0	0
BTC	0	0.5	0.5	0
DVF	0	0.5	0.5	0
RMF	0	0	1	0
RS	0.5	0.5	0	0
TTI	0	1	0	0
VC	0	0	1	0
VFH	0	0	1	0

(b)

Metric	x1	x2	x3	x4
BD	0	0	1	0
BTC	0.5	0	0	0.5
DVF	0.5	0	0	0.5
RMF	0.5	0.5	0	0
RS	0	0.5	0.5	0
TTI	0.5	0	0.5	0
	$0.\dot{3}$	0	$0.\dot{3}$	$0.\dot{3}$
	0.25	0.25	0.25	0.25
VC	1	0	0	0
VFH	0.5	0	0	0.5
	0	0.5	0	0.5
	$0.\dot{3}$	0	$0.\dot{3}$	$0.\dot{3}$

Table 3.10 Optimisation of the ideal mixture for (a) maximum and (b) minimum flammability for litter from Rofe Park for BD-bulk density; TTI – time to ignition; VFH – visual flame height; RS – rate of spread; VC – volume consumed; RMF – residual mass fraction; BTC – burn to completion; DVF – duration of vertical flame; x1 – *Allocasuarina littoralis*; x2 – other; x3 – twigs; x4 – leaves; x5 – decomposed.

(a)

Metric	x1	x2	x3	x4	x5
BD	0	0	0	0	1
BTC	$0.\dot{3}$	$0.\dot{3}$	$0.\dot{3}$	0	0
DVF	$0.\dot{3}$	$0.\dot{3}$	$0.\dot{3}$	0	0
RMF	$0.\dot{3}$	$0.\dot{3}$	0	$0.\dot{3}$	0
RS	0.25	0	0.25	0.25	0.25
TTI	0	$0.\dot{3}$	$0.\dot{3}$	0	$0.\dot{3}$
VC	$0.\dot{3}$	0	$0.\dot{3}$	$0.\dot{3}$	0
VFH	$0.\dot{3}$	0	$0.\dot{3}$	$0.\dot{3}$	0

(b)

Metric	x1	x2	x3	x4	x5
BD	$0.\dot{3}$	0	$0.\dot{3}$	$0.\dot{3}$	0
BTC	0	0.25	0.25	0.25	0.25
DVF	0	0.25	0.25	0.25	0.25
RMF	0	0	1	0	0
RS	0	0	1	0	0
TTI	0.5	0.5	0	0	0
	0.5	0	0	0	0.5
	$0.\dot{3}$	$0.\dot{3}$	$0.\dot{3}$	0	0
	$0.\dot{3}$	$0.\dot{3}$	0	$0.\dot{3}$	0
	$0.\dot{3}$	0	0	$0.\dot{3}$	$0.\dot{3}$
VC	0	0	1	0	0
VFH	0	1	0	0	0

Table 3.11 Comparison of visual flame height (VFH) for samples from Halls Creek (H) and Rofe Park (R) with and without C – *Allocasuarina littoralis* (C). HR – percentage change in VFH values for samples from Halls Creek compared to Rofe Park; RRC – percentage change in VFH values for samples from Rofe Park with and without RC – *Allocasuarina littoralis*.

H	R	RC	H	R	RC	HR	RRC
Sample	**Sample**	**Sample**	**VFH (cm)**	**VFH (cm)**	**VFH (cm)**	**% change**	**% change**
HO	RO	RCO	22	3	27	733	11
HT	RT	RCT	2	10	33	20	30
HL	RL	RCL	30	15	30	200	50
HD	RD	RCD	0	5	31	0	16
HOT	ROT	RCOT	16	14	24	114	58
HOL	ROL	RCOL	24	16	38	150	42
HOD	ROD	RCOD	1	0	29	0	0
HTL	RTL	RCTL	20	18	43	111	42
HTD	RTD	RCTD	1	15	18	7	83
HLD	RLD	RCLD	12	15	41	80	37
HOTL	ROTL	RCOTL	8	20	23	40	87
HOTD	ROTD	RCOTD	3	12	12	25	100
HOLD	ROLD	RCOLD	1	11	28	9	39
HTLD	RTLD	RCTLD	0	6	26	0	23
HOTLD	ROTLD	RCOTLD	11	4	23	275	17

3.4 Discussion

This study is unique because it is the first time the SCD has been used to test the optimum mixtures of fuel fractions. This established chemistry mixture design enables a wider composition mix to be used so it is more representative of surface litter that is burnt in forests. Decomposed materials, twigs, leaves, woody fruits and bark can be found in surface litter in most forests so this experimental method can be applied to other forest types and different plant species as well.

It is also the first time a GBM was used to determine the best statistical model fit for flammability metrics (ignitability, combustability, consumability and sustainability). Flammability measures included time to ignition, burn to completion, visual flame height, rate of spread, volume consumed, and residual mass fraction. The data were optimised to find the maximum or minimum fit for the SCD.

3.4.1 Fuel hazard and flammability

Vegetation found at both Halls Creek and Rofe Park sites are classified as Sydney Coastal Dry Sclerophyll Forest (Keith 2004). From Section 2.3 it was established that Halls Creek and Rofe Park had similar fuel hazard scores (FHS) determined by surface litter depth (i.e. the highest rating of 4) indicating that they were both at extreme risk of bushfire. However, flammability measurements demonstrate considerable differences in litter flammability between these sites. Both sites had rapid time to ignition (TTI) measurements. For Halls Creek, 20% of the mixtures had TTI of 1 s, 7% were 2 s and 7% were 3 s (Table 3.5). For Rofe Park, 16% of the mixtures had TTI of 1 s, 16% were 2 s and 26% were 3 s (Table 3.6). The rate of spread (RS) was twice as rapid for Rofe Park litter mixtures than for Halls Creek (Table 3.5). Similarly, 60% of samples from

Halls Creek had no RS (burning was not sustained) in comparison with 35% from Rofe Park (Table 3.5 and Table 3.6, respectively). For visual flame height (VFH), the results were variable when comparing samples from Halls Creek and Rofe Park with the same litter components (Table 3.11). However, when *Allocasuarina littoralis* was mixed with litter from Halls Creek with the same proportions of components as for Rofe Park samples, VFH was consistently positive with an increase of 11 to 100% (Table 3.11). *Allocasuarina littoralis* had a non- additive effect when included in litter mixtures for RS, VFH, DVF, BTC and VC (Table 3.8) making the litter more flammable and, potentially, the site more flammable overall.

3.4.2 Comparison of methods

Experimental methods of this study and the studies of Della Rocca *et al*. (2018) and de Maghalães and Schwilk (2012) can be compared because the flammability of mixtures was determined. Different experimental methodologies were used in this study and the studies of Della Rocca *et al*. (2018) and de Maghalães and Schwilk (2012) to measure flammability of vegetation and yet non-additive effects were found in all cases. For this study, litter mixtures were arranged in a small round heatproof container and ignited using a cotton ball soaked in ethanol. This was different to the overall method used by Della Rocca *et al*. (2018) using a mass loss calorimeter and by de Maghalães and Schwilk (2012) using a propane torch with litter arranged on a burn table with larger dimensions than used in the current study. The experimental methodologies of Della Rocca *et al*. (2018) and de Maghalães and Schwilk (2012) had litter and leaves that were specified by plant species. In contrast, litter used in this research was arranged according to litter components including twigs, leaves, the decomposed fraction and all other components grouped together (e.g. bark, woody fruits) which may have originated from a number of species. The only component with the plant species specified was cladodes of *Allocasuarina littoralis*. However, to compare and contrast these studies, the litter components with the lowest bulk densities and largest leaf length can be examined. For Halls Creek, the litter component with the lowest bulk density was leaves, while for Rofe Park this was *Allocasuarina littoralis*. Cladodes from this species had the greatest positive influence on RS, twigs and the decomposed fraction had the most positive influence on TTI. Similarly, for litter from Halls Creek, the combined 'other' fraction and twigs had the greatest influence on RS and twigs had the greatest influence on TTI.

Della Rocca *et al*. (2018) found that leaves with the largest surface area drove flammability but de Maghalães and Schwilk (2012) found the species with the longest leaves were crucial. As individual species were not examined in the current study, neither of these patterns can be confirmed, however, the fractions with lowest bulk density (leaves and *Allocasuarina* cladodes) drove flammability. Similarly, previous studies have found that RS had a positive relationship with large leaf area. For example, species that burn the most rapidly include *Quercus kelloggii* (Black oak) and *Pinus ponderosa* (Ponderosa pine) (de Maghalães and Schwilk 2012).

Not only were different methods used to determine flammability, most of the flammability components in the studies available used different units of measurement making comparisons difficult. Two exceptions were TTI and residual mass fraction (RMF). The fastest TTI for this study was 1 s for certain litter mixtures from both Halls Creek and Rofe Park (Table 3.5 and Table 3.6, respectively). In contrast, the fastest TTI for *Quercus faginea* was 2.7 s and 1.2 s for *Quercus kelloggii* and 1.9 s for *Abies* *concolor* (Della Rocca *et al*. 2018).

The litter mixtures used in this study may be considered to be more flammable than material used in other studies because the RMF values were lower compared to studies of Della Rocca *et al*. (2018) and de Maghalães and Schwilk (2012). Litter mixes from both Halls Creek and Rofe Park had RMF values (reported as mass loss (%)) ranged from as low as 0.14% (HOT; Table 3.5) to values as high as 9.09% (ROTL; Table 3.6) and 10.58% (RCOTLD; Table 3.6). In contrast, the lowest RMF of 7.3% was found for *Quercus ilex* subsp. *ilex* and *Juniperus thurifera* (Della Rocca *et al*. 2018). Similarly, the lowest RMF for *Pinus ponderosa* was 3.2% and for *Abies magnifica* was 3.3% (de Maghalães and Schwilk 2012).

3.4.3. Flammability components

There were individual litter components that had a significant positive effect on flammability metrics in litter mixes but they were not the same components for both sites. For example, for Halls Creek, twigs positively affected TTI and the other fraction and leaves both positively affected VFH (see Table 3.7 for other examples). In contrast, there were a number of individual components in litter from Rofe Park that had a very strong positive effect but the most common one was *Allocasuarina littoralis* including RS, VFH, RMF and VC (see Table 3.8 for more examples).

Allocasuarina littoralis was a component that individually had a very strong positive effect on flammability metrics and also a strong effect on interactions. *Allocasuarina littoralis* had a non-additive effect on five flammability metrics including BTC, RS, VC, VFH and DVF. This component was very strong on its own and had an even stronger effect on mixtures of two flammability metrics, for example, the combination of

Allocasuarina littoralis cladodes, leaves and the decomposed fraction had a strong effect on BTC and the combination of *Allocasuarina littoralis* cladodes and leaves had a strong effect on DVF (Table 3.8).

3.4.4. Additive and non-additive effects

In this study, an established chemistry design metholodology, the SCD, was used to find the optimal flammability effect from individual components and for the mixtures. Determining whether non-additive effects occurred in other studies (e.g. de Maghalães and Schwilk 2012; Della Rocca *et al*. 2018) involved quite different methods. The optimal flammability mixture equation for each flammability metric was found by using the GBM of Brown *et al.* (2015) with non-additive effects for each flammability metric based on the lowest AICc value. These non-linear equations demonstrated both positive and negative effects with *Allocasuarina littoralis* enhancing flammability for several of the metrics measured. Della Rocca *et al*. (2018) and de Maghalães and Schwilk (2012) used alternative methods to determine non-additive effects. In the study of de Maghalães and Schwilk (2012), leaf litter with single species measurements were used to determine a weighted mean as a null expectation (see Chapter 1). Any value that was different to this null measurement was considered to be a non-additive effect. Similarly, Della Rocca *et al*. (2018) used the arithmetic sum of the effects of each component species in monospecific fuel to find the expected flammability value (see Chapter 1). Any value that was different to this expected value indicated a non-additive effect.

3.4.5. The importance of shape and size of litter

Leaf shape and size has been found to be important for accounting for non-additive flammability effects of mixtures (de Maghalães and Schwilk 2012; Della Rocca *et al*. 2018). In this study, *Allocasuarina littoralis*, having long needle-like leaves, similar to pines, drove the flammability of litter mixtures from Rofe Park. Similarly, the species, *Pinus lambertuiana*, *P. jeffreyii*, *P. ponderosa* and *Quercus kelloggii* provided non-additive effect to litter mixtures (de Maghalães and Schwilk 2012) and species of *Quercus* with long leaves dominated flammability of the mixtures (Della Rocca *et al*. 2018).

Leaves from native Australian plant species are generally of a different shape and size (i.e. longer and slimmer) than leaves from the species studied elsewhere for flammability (de Maghalães and Schwilk 2012; Della Rocca *et al*. 2018). This could explain the faster TTI and RS and differences in other flammability metrics that have been reported here. However, the species of *Pinus* and *Quercus* tested in other studies also had longer leaf lengths like *Allocasuarina littoralis* that drove the flammability of the mixtures.

The plant species with the longest thinnest leaves seemed to drive flammability in terms of RS and VFH. For example, in this study, *Allocasuarina littoralis* in litter mixtures had VFH of over 40 cm and the most rapid rate of spread 0.67–0.74 cm^{-1}. In the study of de Maghalães and Schwilk (2012), the species with the greatest VFH and RS were *Pinus jeffreyi* and *P. ponderosa*. Other studies have also demonstrated that leaves with similar shape ignite rapidly, burn quickly and are hotter with greater flame heights (Scarff and Westoby 2006; Kane *et al.* 2008).

3.4.6 Using flammability in land management

Land managers need to have information about fuel loads and flammability to help guide them in mitigation of risk from bushfire. For example, in the Bush Fire Risk Management Plan 2016–2021, the Hornsby/Ku-ring-gai Bush Fire Management Committee identified and ranked community asset types at risk of bushfires on the basis of fire history, ignition cause and patterns (see Chapter 1). To do this they use information from a diverse range of sources including Rural Fire Service guidelines, vegetation classes of Keith (2004) and information about fuel loads, local knowledge and fire history, including past prescribed burns. Having information about flammability metrics such as TTI, RS and VFH could also be used to assist them to prioritise where and when to conduct prescribed burns, particularly since, as shown in this study, the vegetation classes of Keith (2004) are broad and include forests with different fire behaviour. Vegetation classifications could be refined with information about flammability of forests. If there is a plant species that is known to be highly flammable in terms of a positive non-additive effect on litter that has a high fuel load and is near strategic assets, then this information will be useful for land managers. For example, flammability metrics related to Rofe Park (e.g. RS and VFH) were different for samples from Halls Creek. Since both sites are on urban-bush interface and have both been classed as extreme risk of fire, the more flammable site is Rofe Park. This suggests that doing prescribed burns in and around Rofe Park should be the higher priority than for Halls Creek.

One of the hypotheses of this study was that the composition of surface litter is a key control of flammability in Sydney Coastal Dry Sclerophyll Forest. There is a non-additive effect with the most flammable component of litter determining flammability of the litter mixture. The evidence from this research suggests the hypothesis is correct based on experimental design and optimal flammable effect since *Allocasuarina littoralis* had a significant positive non-additive effect on several flammability metrics.

Optimisation of flammability metrics and litter components can assist land managers in determining which components are a bigger risk when they are on their own or mixed in the litter bed. Land managers must rank assets in terms of whether they are a low, medium or high risk of bushfire. For example, Rofe Park was identified as a particularly flammable site based on flammability measurements. *Allocasuarina littoralis* in mixtures produced maximum values optimum values for RS, VFH, DVF and BTC. In comparison, the decomposed fraction had a negative non-additive effect on litter mixtures in both Halls Creek and Rofe Park. Therefore, if a site has no *Allocasuarina littoralis* and a high proportion of decomposed litter then it could be at a low risk of bushfires.

Halls Creek and Rofe Park represent the same forest classification of Sydney Dry Coastal Sclerophyll Forest (Keith 2004) but this study has demonstrated different flammability characteristics. Even though TTI was rapid in litter mixtures for both sites, a high proportion of litter mixes from Halls Creek failed to spread whereas several mixtures from Rofe Park had a rapid RS and high values for VFH. This suggests that bushfire may spread more rapidly through the forest at Rofe Park compared to Halls Creek and potentially pose a greater risk to strategic assets on the urban-bush interface.

Chapter Four: General conclusions and management implications

4.1 Introduction

Because fire shapes vegetation globally, a fundamental goal of functional ecology is to scale from plant traits to ecosystem effects (Schwilk 2015). The mechanistic basis for scaling has been elusive because previous studies from laboratories, mathematical models and field tests have produced inconsistent results (Fernandes and Cruz 2012). White and Zipperer (2010) demonstrated that there are many different methodologies for testing flammability (see Table 1.7). Land managers of fire-prone ecosystems need empirical data to describe the patterns and mechanisms of flammability in an ecological context (Varner *et al*. 2015). This study has provided empirical data of the patterns and mechanisms of the flammability of forest litter from two sites at extreme risk of bushfire by combining laboratory experiments with a chemistry mixture design and a mathematical model to determine the flammability of litter components and interactions between them. The Simplex Centroid Design (SCD) was used to find the optimal flammability of litter components and mixtures. The General Blending Model (GBM) was used to determine equations with the best statistical fit. The non-linear equations demonstrated either positive or negative non-additive effects of a litter component influencing fire behaviour of the litter mixture. These equations were optimised to find litter mixtures with the minimum and maximum flammability. This is useful for land managers because litter components with the lowest and highest flammability in an ecosystem can be determined and taken into account when planning prescribed burns.

Study sites at Halls Creek and Rofe Park were chosen as being representative of long unburnt Sydney Coastal Dry Sclerophyll Forest (Keith 2004). As described in Chapter 2, both sites had similar vertical structure, litter depth and fire hazard indicating an extreme risk of bushfires. There were ladder fuels in every section of the vertical structures to carry fire into the canopies. Fuel hazard scores of litter fuels were similar but flammability traits were found to be significantly different. The results presented in Chapter 3 demonstrated the presence of *Allocasuarina littoralis* litter at Rofe Park could potentially increase the risk of bushfires because cladodes (modified leaves) from this species enhanced flammability of litter mixtures with a positive non-additive effect. Twigs were the most flammable component in the litter from Halls Creek and *Allocasuarina littoralis* cladodes were the most flammable component in litter from Rofe Park. This study therefore supported the hypothesis that composition of surface litter is a key control of flammability in Sydney Coastal Dry Sclerophyll Forest.

Land managers can use information about fuel loads and flammability for guidance in mitigation of risk from bushfires. Fuel loads and fuel hazard scores indicate the amount of litter that is present as fuel for bushfires (Chapter 2). The flammability of litter provides information about fire behaviour to provide a better understanding of how a site may burn (Chapter 3). Land managers rank sites at risk of bushfires into low, medium, high, very high and extreme to prioritise prescribed burns. Information about flammability of litter components at the sites assists with this classification.

4.2 Applicability of flammability conditions and predictions

There are many different types of Dry Sclerophyll Forest with features that are specific to the topography, geology, climate and geographical range in their locality (Keith

2004). The common features of Dry Sclerophyll Forest are an open canopy of sclerophyllous trees with a lower conspicuous layer of sclerophyllous shrubs (Keith 2004). Sydney Coastal Dry Sclerophyll Forests are the most biodiverse and extensive in Australia and are unique to the greater Sydney Basin (Keith 2004). Using sites from this particular forest type for flammability studies will allow knowledge to be transferrable to other categories of Dry Sclerophyll Forest found throughout the Sydney Basin and further afield in NSW. However, there are idiosyncrasies in the results, but these are related to surface litter at the two sites. These results though have to be viewed within the wider description and context of the sites, for example, slope, rural-bush or urban-bush interfaces, wind, aspect and other factors to give an overall hazard assessment. In addition, the high R^2 values means there are effectively little or no residuals in the model when fitted against the experimental dataset. As the GBM is a statistical model, its use attempts to maximise the fit (hence the R^2) by changing the value of the different exponents. This is an artefact of the approach, however, the quality of the model adopted was assessed by using AICc values. Furthermore, the SCD covers a wide range of mixtures of litter for a given vegetation class. In theory, the resulting flammability metrics should also encompass a wide range of possible responses within the vegetation class providing the land manager with confidence that at another site of the same vegetation class that the potential flammability response could be predicted, as long as all other parameters were equal. This application obviously would need testing against material collected from another site.

The experimental methodology of this study attempted to replicate ignition of forest litter in the field with the cotton bud as a substitute for an ember or application of liquid fuel during manual ignition (Pluckinski and Anderson 2008 and see Section

3.2.1). Although the flammability study was conducted in a laboratory setting it is applicable to field conditions and can be used for interpretation of fire risk by land managers.

Currently, the assumption about assessing bushfire risk associated with forest litter is the same for all forest types and is only dependent on the depth and connectivity of the litter. According to fuel assessment guides used by land managers, any very thick layer (>25 mm) of completely connected litter is the greatest bushfire risk (Hines *et al.* 2010; Gould *et al*. 2011). However, this does not explain patchiness of fires that are common with prescribed burns in dry sclerophyll forest in south-eastern Australia or bushfires, such as the Mount Hall and Nattai bushfires (Chafer 2004; Penman *et al*. 2007; Bradstock *et al.* 2010). It is well known that fires are commonly of lower intensity in gullies and sheltered areas that can be moister or because vegetation is less flammable, or both (Chafer 2004; Penman *et al.* 2007; Bradstock *et al.* 2010). This is regardless of litter depth and presumed fire risk afforded by fuel hazard guides. As such, optimum maximum and minimum flammability limits were calculated and presented in this study (see Chapter 3, Tables 3.9 and 3.10, respectively). However, as all of the litter used in the study had been oven-dried and stored in sealed containers (see Section 2.2.4), the experimental results of this study represent the 'worst case scenario' as they were in a state representing the 'perfect' burn under 'perfect' conditions. Della Rocca *et al.* (2018) and de Maghalães and Schwilk (2012) oven-dried their samples as well so it is the mathematical method that produces different and more meaningful results. The SCD is for the optimal result to be identified, which in this case was for flammability. The other studies did not necessarily find the optimal result as they were limited by their experimental methodology with fewer litter components in their mixtures.

There were significant differences in values of flammability variables with litter from Halls Creek and Rofe Park, the latter with and without cladodes from *Allocasuarina littoralis*. For example, 60% of Halls Creek samples did not sustain burning (no rate of spread) in comparison to 35% from Rofe Park, even if values for time to ignition were rapid (see Chapter 3, Tables 3.5 and 3.6, respectively). *Allocasuarina littoralis* litter mixtures had the greatest visual flame heights and most rapid rates of spread (see Tables 3.6 and Table 3.11, respectively). *Allocasuarina littoralis* also had a positive non-additive effect in litter mixtures for other flammability variables including: duration of vertical flame, burn to completion and volume consumed, making the site more flammable (Table 3.8). It is most likely that *Allocasuarina littoralis* had a non-additive effect on flammability of litter because of long thin leaves. This result is consistent with other experimental studies that found species with long or larger leaves are more flammable because they ignite and burn rapidly, while producing tall hot flames with a higher intensity and faster rate of spread (Rothermel 1972; Scarff andWestoby 2006; Kane *et al.* 2008; de Maghalães and Schwilk (2012).

In this study several caveats need to be taken into account when applying the laboratory results to the field. During burning experiments there were none of the common features associated with forest litter such as variable fuel moisture, wind, slope, aspect and topography. *Allocasuarina littoralis* litter was dry apart from a few centimetres above the soil where it was very moist, which would affect flammability. This was not well represented by using oven-dried samples. Similarly, aspect will have a role in how wet or dry the litter is *in situ*.

At both sites there was a constant breeze because of the weather and from different levels of vegetation on the steep slopes. Wind is such a common feature in eucalypt forests that 'The McArthur Forest Danger Meter' has been criticised for substantially under-predicting rate of spread under high wind speeds (Noble *et al.* 1980; Bradstock *et al.* 2010). Weather can have a greater influence on area burned during a bushfire than fuel load (Boer *et al.* 2008; Bradstock *et al.* 2010). The Halls Creek and Rofe Park sites were located in densely forested steep terrain. The role of slope is well known as fire moves rapidly up steep slopes and burn more intensely on plateaux and less intensely in moist gullies causing patchiness of burns (Bradstock *et al.* 2010). All of these features will affect litter flammability in the field, but this does not preclude laboratory-based information being used by land managers to inform their decision-making processes.

4.3 Implications for land management

It is recommended that the experimental methodology using the SCD, GBM and optimisations, be used in conjunction with existing land management methods. The topography, boundaries and flammability of litter associated with Halls Creek and Rofe Park sites have different implications for land management and must also be taken into account by land managers. In this study, the Halls Creek site was found to be not particularly flammable in the small section investigated but is at the urban-bush interface. The area of the Halls Creek site is relatively small, about 750 x 530 m in area, but it has a t-shape about 10–20 m at the end and is surrounded by large areas of bushland connected to Berowra Valley National Park (pers. comm. Brown and Jones 2016). Access to the site is at the end of a street and through a private rural property. In addition to poor accessibility (see Chapter 2, Section 2.2.1) (pers. comm. Brown and Jones 2016), the terrain is rugged and steep (100–220 m altitude) and densely forested in places. The flammability of the vegetation in the ecotone at the end of the site is unknown. All of these factors add to the risk associated with the vegetation in this area.

According to the flammability study presented here, Rofe Park has highly flammable litter. In addition, Rofe Park is next to several community assets that increase the bushfire risk rating (see Chapter 3, Section). It has an area of about 435 x 370 m including a park and sportsgrounds clear of vegetation (pers. comm. Brown and Jones 2016). This site is on an urban-bush interface surrounded by many residential properties and a small shopping centre with a petrol station. Other nearby assets include Hornsby Heights Community Centre, Hornsby Dog Training Club and Hornsby North Public School (pers. comm. Brown and Jones 2016). Accessibility to the site is excellent since it is off the street and at the end of a cul-de-sac (pers. comm. Brown and Jones 2016). In addition, it is a site of local significance with protected species including koalas, Powerful Owls and Red-crowned Toadlets (pers. comm. Brown and Jones 2016). On the broader scale, both the Halls Creek and Rofe Park sites have forests that are connected to Berowra Valley National Park and are also connected to Ku-ring-gai National Park. Both sites are examples of how the surrounding context needs to be considered as part of the fire risk of forested areas, particularly when the urban-bush or rural-bush interface is as evident as it is in the Hornsby Shire.

In north-eastern USA, fire suppression policies of fire-prone pine forests with dense populations on urban-bush interfaces increase tree density and fuel loads which promotes greater fire intensity, severity and increased probability of crown fires (Taylor 2004; Hutchinson *et al.* 2008; Nowacki and Abrams 2008; Ryan *et al.* 2013). There are also pine plantations in south-eastern Australia that provide more flammable fuel although the overall extent of *Allocasuarina* and *Casuarina* is quite small (Carnegie *et al.* 2005; Fairley and Moore 2010). In areas at risk of bushfires, suppression of fires near homes is not a sufficient form of land management because prescribed burning or mechanical clearing and other land management strategies specified in the Bush Fire Risk Management Plan are still required to reduce the overall risk of fire (Hornsby/Ku-ring-gai Bush Fire Risk Management Plan 2016-2021).

4.4 Recommendations and further research

It would be ideal to have standard experimental methodologies for flammability studies of different vegetation types in Australia and throughout the world (White and Zipperer 2010; Varner *et al*. 2015). The methodology used in this thesis is not restricted to Sydney Coastal Dry Sclerophyll Forest and can be applied to other vegetation types. To investigate the suitability of modelling methods described in this study, it would be useful to test the flammability of litter from other sites at extreme risk of bushfires such as locally in the Blue Mountains, NSW or in other states such as Victoria. Litter could first be tested under laboratory conditions and then tested in the field under controlled conditions such as in Project Vesta (Gould *et al.* 2008). This last step would be costly and time consuming but has been achieved on other occasions. The same experiments can also be done with different moisture contents, for example, at 10% and 30% fuel moisture content, reminiscent of field conditions, in a fully-crossed mixture experiment design. This would allow for examination of the flammability of litter mixtures that have different levels of moisture content, as is commonly found in the field in moist gullies and on dry ridges or plateaux (Bradstock *et al*. 2010).

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Appendix

Table A1 Gross heat of combustion (GHC) of samples from Halls Creek (H) according to bomb calorimetry analysis. Transect number is indicated by the first number (1, 2 or 3), multiples of five are the positions along the transect (m). St. dev. – standard deviation.

Sample ID	**Weight (g)**	**GHC (MJ kg^{-1})**
H15	0.7717	18.6081
H115	0.7944	18.3785
H125	0.7663	16.9292
H135	0.7636	18.4094
H145	0.7580	18.4213
H25	0.7852	18.5110
H215	0.7810	18.7783
H225	0.7701	17.7804
H235	0.7585	18.8950
H245	0.7780	18.3716
H35	0.7858	19.0267
H315	0.7658	18.6902
H325	0.7720	18.5939
H335	0.7718	18.6235
H345	0.7740	18.5417
Mean		18.4373
St. dev.		0.5040

Table A2 Gross heat of combustion (GHC) of samples from Rofe Park (R) according to bomb calorimetry analysis. Transect number is indicated by the first number (1, 2 or 3), multiples of five are the positions along the transect (m). St. dev. – standard deviation.

Sample ID	**Weight (g)**	**GHC ($MJ\ kg^{-1}$)**
R15	0.7685	18.5196
R115	0.7565	17.8126
R125	0.7541	18.5597
R135	0.7507	18.1207
R145	0.7700	18.8528
R25	0.7486	17.5215
R215	0.7521	17.4140
R225	0.7478	18.9552
R235	0.7712	18.2774
R245	0.7456	17.7477
R35	0.7528	18.6672
R315	0.7612	16.7593
R325	0.7765	17.4113
R335	0.7414	17.8489
R345	0.7596	17.9814
Mean		18.0300
St. dev.		0.6168

Table A3 Net heat of combustion (NHC) of samples from Halls Creek (H) according to bomb calorimetry analysis. Transect number is indicated by the first number (1, 2 or 3), multiples of five are the positions along the transect (m). St. dev. – standard deviation.

Net heat of combustion = Gross heat (MJ kg^{-1}) – (Latent heat of vaporisation of water (MJ kg^{-1}) × (H_2O/H) × (Hydrogen %/100)

Where latent heat of vaporisation of water = 2.256 MJ kg^{-1}, Water molar mass = 18.016 g mol^{-1}, Hydrogen molar mass =1.008 g mol^{-1}, H_2O/H = 17.873.

Sample ID	Gross heat (MJ kg^{-1})	Hydrogen (%)	NHC (MJ kg^{-1})
H15	18.6081	6.4	16.0154
H115	18.3785	6.2	15.8850
H125	16.9292	6.0	14.5111
H135	18.4094	6.6	15.7333
H145	18.4213	6.3	15.8996
H25	18.5110	6.3	15.9909
H215	18.7783	6.5	16.1473
H225	17.7804	6.4	15.1978
H235	18.8950	6.3	16.3358
H245	18.3716	5.8	16.0140
H35	19.0267	6.5	16.3965
H315	18.6902	6.5	16.0915
H325	18.5939	6.4	15.9952
H335	18.6235	6.5	15.9982
H345	18.5417	6.4	15.9813
Mean		6.3	15.8795
St. dev.		0.2	0.4658

Table A4 Net heat of combustion (NHC) of samples from Rofe Park (R) according to bomb calorimetry analysis. Transect number is indicated by the first number (1, 2 or 3), multiples of five are the positions along the transect (m). St. dev. – standard deviation.

Net heat of combustion = Gross heat (MJ kg^{-1}) – (Latent heat of vaporisation of water (MJ kg^{-1}) × (H_2O/H) × (Hydrogen %/100)

Where latent heat of vaporisation of water = 2.256 MJ kg^{-1}, Water molar mass = 18.016 g mol^{-1}, Hydrogen molar mass =1.008 g mol^{-1}, H_2O/H = 17.873.

Sample ID	**Gross heat (MJ kg^{-1})**	**Hydrogen (%)**	**NHC (MJ kg^{-1})**
R15	18.5196	6.2	16.0172
R115	17.8126	6.3	15.2615
R125	18.5597	6.6	15.8787
R135	18.1207	6.6	15.4458
R145	18.8528	6.8	16.1303
R25	17.5215	6.2	15.0280
R215	17.4140	6.7	14.7088
R225	18.9552	6.7	16.2387
R235	18.2774	6.6	15.6335
R245	17.7477	6.5	15.1276
R35	18.6672	6.3	16.1463
R315	16.7593	6.2	14.2473
R325	17.4113	6.0	14.9908
R335	17.8489	6.5	15.2191
R345	17.9814	6.4	15.4020
Mean		6.4	15.4317
St. dev.		0.2	0.5784

Table A5 Nitrogen, carbon and hydrogen composition (%) of samples from Halls Creek (H). Transect number is 1, 2 or 3, multiples of five are the position on the transect (m). St. dev. – standard deviation.

Name	Mass (mg)	Nitrogen (%)	Carbon (%)	Hydrogen (%)
H15	53.1	0.7	46.7	6.4
H115	50.4	0.8	47.9	6.2
H125	53.2	0.7	45.7	6.0
H135	52.6	0.6	47.2	6.6
H145	51.4	0.8	47.5	6.3
H25	50.1	0.6	47.0	6.3
H215	52.0	0.7	47.9	6.5
H225	54.0	0.7	45.0	6.4
H235	54.3	0.8	46.3	6.3
H245	50.6	0.7	48.0	5.8
H35	56.6	0.6	48.7	6.5
H315	58.1	0.6	48.9	6.4
H325	51.7	0.8	48.0	6.4
H335	52.2	0.5	46.8	6.5
H345	58.3	0.7	48.2	6.4
Mean	53.2	0.7	47.3	6.3
St. dev.	2.6	0.1	1.1	0.2

Table A6 Nitrogen, carbon and hydrogen composition (%) of samples from Rofe Park (R). Transect number is 1, 2 or 3, multiples of five are the position on the transect (m). St. dev. – standard deviation.

Name	**Mass (mg)**	**Nitrogen (%)**	**Carbon (%)**	**Hydrogen (%)**
R15	50.0	1.2	47.4	6.2
R115	50.0	1.0	45.5	6.3
R125	57.2	1.1	47.3	6.6
R135	51.9	1.0	46.2	6.6
R145	52.2	1.0	46.9	6.8
R25	51.5	0.7	42.2	6.2
R215	53.4	1.0	46.0	6.7
R225	51.3	1.2	48.0	6.7
R235	51.0	0.8	44.0	6.6
R245	52.9	0.9	45.2	6.5
R35	50.1	0.6	47.0	6.3
R325	52.2	0.8	45.4	6.2
R315	53.3	1.0	44.0	6.0
R335	53.6	1.1	46.5	6.5
R345	55.9	1.1	45.4	6.4
Mean	52.4	1.0	45.9	6.4
St. dev.	2.1	0.2	1.4	0.2

Table A7 Summary of fitted (fitted(M)) and residual (resid(M)) values comparing experimental and model amounts for samples from Halls Creek. BD – bulk density; BTC – burn to completion; DVF – duration of vertical flame; RS – rate of spread; RMF – residual mass fraction; TTI – time to ignition; VFH – visual flame height; VC – volume consumed.

		fitted(M)	**resid(M)**
BD	1	25.48	2.52
	2	79.72	-2.72
	3	14.89	0.11
	4	53.75	-0.75
	5	33.24	-2.24
	6	20.19	0.81
	7	39.62	-3.62
	8	26.09	-0.09
	9	66.74	7.26
	10	34.32	2.68
	11	22.36	4.64
	12	37.18	2.82
	13	31.37	-3.37
	14	47.59	-2.59
	15	29.45	-5.45
BTC	1	135.25	66.75
	2	114.93	5.07
	3	192.59	-13.59
	4	125.09	-28.09
	5	163.92	59.08
	6	72.99	-67.99
	7	328.73	6.27
	8	62.83	108.17
	9	101.66	8.34
	10	162.95	-71.95
	11	86.97	-74.97
	12	116.11	2.89

Table A7 continued.

		fitted(M)	resid(M)
DVF	1	138.49	61.51
	2	89.09	2.91
	3	188.40	-16.40
	4	113.79	-22.79
	5	163.44	58.56
	6	70.06	-68.06
	7	321.96	6.04
	8	45.36	101.64
	9	95.01	9.99
	10	154.74	-70.74
	11	76.40	-73.40
	12	107.26	10.74
RS	1	0.25	< 0.01
	2	0.18	< 0.01
	3	0.41	< 0.01
	4	0.19	< 0.01
	5	0.15	< 0.01
	6	0.25	< 0.01
RMF	1	35.80	-14.41
	2	86.74	13.26
	3	62.58	0.30
	4	121.67	-21.67
	5	0.18	-0.04
	6	49.19	-10.48
	7	78.73	21.27
	8	74.66	-29.25
	9	104.20	-4.20
	10	92.12	7.88
	11	89.90	10.10
	12	109.40	-9.40
	13	72.61	27.39
	14	89.42	10.58
	15	100.94	-0.94

Table A7 continued.

		fitted(M)	resid(M)
TTI	1	-1.16	3.16
	2	24.20	3.80
	3	2.07	4.93
	4	11.52	-5.52
	5	0.45	0.55
	6	4.46	-1.46
	7	13.14	-6.14
	8	20.11	3.89
	9	9.04	-4.04
	10	8.37	-1.37
	11	7.58	1.42
	12	0.20	0.80
	13	14.10	0.90
	14	1.92	-0.92
VFH	1	0.21	0.01
	2	0.05	-0.03
	3	0.32	-0.02
	4	0.13	0.03
	5	0.20	0.044
	6	0.05	-0.04
	7	0.19	0.01
	8	-0.03	0.04
	9	0.10	0.02
	10	0.13	-0.05
	11	0.05	-0.02
	12	0.07	-0.06
	13	0.05	0.06
VC	1	< 0.01	< 0.01
	2	< 0.01	< 0.01
	3	< 0.01	< 0.01
	4	< 0.01	< 0.01
	5	< 0.01	< 0.01

Table A8 Summary of fitted (fitted(M)) and residual (resid(M)) values comparing experimental and model amounts for samples from Rofe Park. BD – bulk density; BTC – burn to completion; DVF – duration of vertical flame; RS – rate of spread; RMF – residual mass fraction; TTI – time to ignition; VFH – visual flame height; VC – volume consumed.

		fitted(M)	**resid(M)**
BD	1	6.81	8.20
	2	23.41	4.59
	3	20.93	8.07
	4	20.51	0.49
	5	92.39	2.61
	6	15.11	-4.11
	7	13.87	-2.87
	8	13.66	3.34
	9	12.87	0.13
	10	22.17	-0.17
	11	21.96	2.04
	12	2.40	43.60
	13	3.28	20.72
	14	-2.66	56.66
	15	1.05	30.95
	16	-5.05	17.05
	17	-5.91	16.91
	18	-1.70	15.70
	19	-8.08	16.08
	20	-1.05	20.05
	21	-2.36	12.36
	22	-2.62	21.62
	23	40.40	-13.40
	24	32.70	-4.70
	25	37.06	-3.06
	26	17.92	-0.92
	27	20.38	7.62
	28	17.09	4.91
	29	19.00	-4.00
	30	33.60	15.40
	31	20.45	-1.45

Table A8 continued.

		fitted(M)	resid(M)
BTC	1	62.41	1.59
	2	124.23	-9.23
	3	203.22	-10.22
	4	161.00	9.00
	5	93.32	-8.32
	6	132.81	14.19
	7	152.72	-39.72
	8	109.32	-19.32
	9	140.40	8.60
	10	142.61	3.39
	11	182.11	1.89
	12	116.49	3.51
	13	231.44	2.56
	14	166.04	39.96
	15	114.29	-0.29
	16	192.37	14.63
	17	140.62	18.38
	18	113.61	26.39
	19	134.29	2.71
	20	84.05	2.95
	21	110.38	-18.38
	22	177.70	-42.70
	23	118.67	15.33
	24	99.80	-14.80
	25	119.55	-15.55
	26	44.28	-14.28
	27	67.24	27.76

Table A8 continued.

		fitted(M)	resid(M)
DVF	1	71.41	-10.41
	2	111.76	0.24
	3	172.11	-14.11
	4	168.10	-1.10
	5	91.58	-7.58
	6	121.76	23.24
	7	114.95	-3.95
	8	119.46	-30.46
	9	141.93	4.07
	10	139.93	4.07
	11	170.10	10.90
	12	112.10	-3.10
	13	232.61	0.39
	14	179.11	25.89
	15	116.89	-4.39
	16	199.23	5.77
	17	137.01	19.99
	18	114.27	24.73
	19	150.65	-16.65
	20	65.69	13.31
	21	85.81	-5.81
	22	172.53	-42.53
	23	95.75	35.25
	24	95.25	-14.25
	25	110.34	-12.34
	26	43.80	-21.80
	27	71.38	20.62

Table A8 continued.

		fitted(M)	resid(M)
RMF	1	68.15	-7.26
	2	35.43	-1.66
	3	2.88	0.64
	4	57.93	-7.73
	5	51.79	-5.44
	6	65.04	-4.04
	7	63.04	-6.15
	8	20.52	5.78
	9	4.36	2.15
	10	48.61	10.10
	11	53.84	31.50
	12	25.49	-2.40
	13	38.75	8.60
	14	27.76	-5.96
	15	32.99	9.68
	16	14.72	-5.63
	17	35.46	-6.63
	18	33.60	-11.69
	19	19.82	7.31
	20	21.76	-11.18

Table A8 continued.

		fitted(M)	resid(M)
RS	1	0.58	0.03
	2	0.12	-0.03
	3	0.16	0.03
	4	0.47	0.05
	5	0.66	0.01
	6	0.52	-0.07
	7	0.39	0.03
	8	0.14	0.08
	9	0.30	0.04
	10	0.51	-0.06
	11	0.58	0.01
	12	0.54	-0.07
	13	0.33	-0.10
	14	0.54	-0.04
	15	0.25	-0.05
	16	0.43	0.04
	17	0.19	-0.01
	18	0.50	-0.03
	19	0.58	0.16
	20	0.37	< 0.01

Table A8 continued.

		fitted(M)	resid(M)
TTI	1	2.80	0.20
	2	0.24	2.76
	3	36.58	-1.58
	4	0.68	2.32
	5	23.96	1.04
	6	1.52	-0.52
	7	1.87	0.13
	8	1.74	0.26
	9	-0.57	1.57
	10	3.65	-0.65
	11	0.46	1.54
	12	1.82	1.18
	13	13.03	-1.03
	14	12.32	-1.32
	15	4.00	-3.00
	16	1.24	-0.24
	17	7.37	-5.87
	18	0.46	1.54
	19	2.39	-0.39
	20	-0.15	1.15
	21	2.80	0.20
	22	0.24	2.76
	23	36.58	-1.58
	24	0.68	2.32
	25	23.96	1.04
	26	1.52	-0.52
	27	1.87	0.13
	28	1.74	0.26
	29	-0.57	1.57
	30	3.65	-0.65

Table A8 continued.

		fitted(M)	resid(M)
VC	1	< 0.01	< 0.01
	2	< 0.01	< 0.01
	3	< 0.01	< 0.01
	4	< 0.01	< 0.01
	5	< 0.01	< 0.01
	6	< 0.01	< 0.01
	7	< 0.01	< 0.01
	8	< 0.01	< 0.01
	9	< 0.01	< 0.01
	10	< 0.01	< 0.01
	11	< 0.01	< 0.01
	12	< 0.01	< 0.01
	13	< 0.01	< 0.01
	14	< 0.01	< 0.01
	15	< 0.01	< 0.01
	16	< 0.01	< 0.01
	17	< 0.01	< 0.01
	18	< 0.01	< 0.01
	19	< 0.01	< 0.01
	20	< 0.01	< 0.01

Table A8 continued.

		fitted(M)	resid(M)
VFH	1	0.43	-0.06
	2	0.09	-0.06
	3	0.15	-0.05
	4	0.17	-0.02
	5	0.11	-0.06
	6	0.26	0.01
	7	0.29	0.04
	8	0.30	0.00
	9	0.27	0.04
	10	0.12	0.02
	11	0.13	0.03
	12	0.16	0.02
	13	0.11	0.04
	14	0.14	0.01
	15	0.22	0.02
	16	0.39	-0.01
	17	0.21	0.08
	18	0.41	0.01
	19	0.20	-0.02
	20	0.40	0.01
	21	0.43	-0.06
	22	0.09	-0.06
	23	0.15	-0.05
	24	0.17	-0.02
	25	0.11	-0.06
	26	0.26	0.01
	27	0.29	0.04
	28	0.30	0.00
	29	0.27	0.04
	30	0.12	0.02

Definitions from nloptr.print.options()

(Reference: Steven G. Johnson, The NLopt non-linear optimisation package, http://ab-initio.mit.edu/nlopt)

ftol_rel

possible values: ftol_rel > 0
default value: 0.0

Stop when an optimization step (or an estimate of the optimum) changes the objective function value by less than ftol_rel multiplied by the absolute value of the function value. If there is any chance that your optimum function value is close to zero, you might want to set an absolute tolerance with ftol_abs as well. Criterion is disabled if ftol_rel is non-positive (default).

ftol_abs

possible values: ftol_abs > 0
default value: 0.0

Stop when an optimization step (or an estimate of the optimum) changes the function value by less than ftol_abs. Criterion is disabled if ftol_abs is non-positive (default).

maxeval

possible values: maxeval is a positive integer
default value: 100

Stop when the number of function evaluations exceeds maxeval. This is not a strict maximum: the number of function evaluations may exceed maxeval slightly, depending upon the algorithm. Criterion is disabled if maxeval is non-positive

xtol_rel

possible values: xtol_rel > 0
default value: 1.0e-04

Stop when an optimization step (or an estimate of the optimum) changes every parameter by less than xtol_rel multiplied by the absolute value of the parameter. If there is any chance that an optimal parameter is close to zero, you might want to set an absolute tolerance with xtol_abs as well. Criterion is disabled if xtol_rel is non-positive.

xtol_abs

possible values: xtol_abs > 0
default value: rep(0.0, length(x0))

xtol_abs is a vector of length n (the number of elements in x) giving the tolerances: stop when an optimization step (or an estimate of the optimum) changes every parameter x[i] by less than xtol_abs[i]. Criterion is disabled if all elements of xtol_abs are non-positive (default).