Effects of woody weeds on fuels and fire behaviour in Eastern Australian forests and woodlands

by

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Abstract

Fire is a common feature in most ecosystems in Australia. Much of the native flora is well adapted to occasional fire and recovers over time in a variety of ways. Invasive species or 'weeds' are also a common feature in most Australian ecosystems, particularly in forests and woodlands close to urban settlements. Many invasive species have the potential to recover or recolonise more rapidly following disturbance than native species and may change the fuel load and structure of invaded areas. Invasive species can alter the fuel load and structure providing the fine fuel necessary for initiation and propagation of fire. Woody weeds can also provide elevated biomass to sustain fire and 'ladder fuels' allowing fire to reach the canopy. When both of these elements are considered there is the likelihood of alteration of fire behaviour in weed-infested areas of forests and woodlands. The research described in this thesis aims to investigate the effect of invasive species on fire in woodlands of eastern Australia.

The fuel load, fuel structure and flammability of pristine (non-invaded) Cumberland Plain Woodland (Australian Botanical Garden, Mount Annan, New South Wales, Australia) and adjacent areas invaded with the woody weed, African Olive (*Olea europaea* subsp. *cuspidata*), was assessed and compared. Heavily-invaded areas are comprised of mature trees of African Olive present for more than 15 years, with a continuous canopy and a limited number of species in the understorey were contrasted with areas of 'intermediate' invasion, where immature trees of African Olive were interspersed among a grassy/shrubby matrix, and areas of pristine woodland. Overall, there was an increase in fine fuel loads, vertical distribution, fuel hazard score and flammability in areas densely invaded with African Olive compared to more recently invaded areas and nearby pristine

woodland. The differences in fuel load and vertical distribution of invaded and non-invaded areas are likely to result in changes in fire behaviour and will therefore influence the risk of fire. These data were used to model and test fire behaviour.

The native shrub, Cootamundra Wattle (*Acacia baileyana*) has become an invasive woody weed in Yellow Box/Red Gum Grassy Woodland of Red Hill Nature Reserve in the Australian Capital Territory. Measurements of fuel load, fuel structure and flammability of heavily-invaded, sparsely-invaded and nearby pristine woodland indicated that the presence of Cootamundra Wattle changes the vertical distribution of fine fuels in invaded woodlands and changes the fuel hazard rating, but not the flammability of the fuel. As Cootamundra Wattle is a native Australian species it was not unexpected that there was no alteration in fuel flammability.

Information about the combustion and flammability of invasive and native Australian species is scarce. Morphological, chemical and combustion characteristics related to flammability of fuel were measured using leaves from a range of woody weeds and compared to native Australian plants. Flammability of leaf material was measured using a Mass Loss Calorimeter and an Oxygen Bomb Calorimeter while morphological leaf traits and chemical analyses followed well-recognised methods. There was little evidence supporting correlation between leaf morphology and leaf flammability. A novel computational method was used to combine the four distinct components of leaf flammability (ignitability, sustainability, combustibility and consumability) to rank the species tested. The usefulness and limitations of such a ranking system to support fire management decisions is discussed.

Many studies, including the two case studies presented here, have shown the effects of invasive plants on fire behaviour. To date no Australian studies have used fuel data collected from the field with fire behaviour models to predict fire behaviour in areas invaded by woody weeds compared to non-invaded areas. In this study, the parameters required for modelling fuel in invaded and non-invaded vegetation located in the Australian Botanical Garden, Mount Annan were defined. Fire behaviour was simulated in invaded and non-invaded vegetation using the BehavePlus Fire Modeling System and compared to the predictions of fire behaviour using models currently in use by Australian fire management authorities for grassland and forest fuels.

Woody weed invasion in Australian ecosystems are likely to be unique in the way that fuel loads, distribution and hazard ratings are altered. The flammability of invasive species should also be considered as an important variable influencing fire behaviour. Fire behaviour in novel fuel types can be modelled using field data. However, considerable field experimentation is still required to validate our understanding of how woody weeds may alter fire behaviour in different situations.

Conducting research on invasive species and their potential effects on fuel composition and fire behaviour is becoming imperative given the increasing pressure of further woody weed invasions and increased extreme fire weather due to anthropogenic global warming.

For the ones I love

"Set your life on fire

Seek those who fan your flames"

Rumi

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1. Introduction

1.1. Background to research

Australia is recognised worldwide, not only for the prevalence of bushfires, but also for many detrimental examples of biological invasion. Invasive plant species or weeds are able to spread into forests and woodlands surrounding populated areas and, once established, can directly alter ecosystem dynamics. A number of studies have investigated the ecological effects caused by the presence of weeds, however, the effects of invasive plants on fuel load and structure, fire regimes and intensity remains poorly understood, especially in Australia.

It is likely that weeds will alter flammability when compared to native species due to various aspects related to their architecture, leaf morphology, chemistry and life history. Because of the potential to improve management of invasive species and preserve ecosystems threatened by invasion, as well as to elucidate principles of population and community ecology underlying invasion, understanding the relationships among fire, plant invasion and plant community structure is currently of great interest to scientists and managers. Investigation of the alteration of fuel and the flammability of the most problematic weeds in Australia will allow the development of specific fuel models for these novel fuels. Using these fuel models as input to fire behaviour prediction systems could potentially:

1. Improve the management and application of fuel reduction fires,
2. Support better responses to fire emergencies in weed-invaded areas, and

3. Ultimately, result in best management practices to control weeds and reduce risk to the community.

1.2. Thesis structure

Chapter 1 – A broad review of the most important aspects of plant invasion in Australia in relation to fire behaviour is presented. Fire behaviour in the main types of vegetation is described and an introduction to fire prediction models and modelling is provided.

Chapter 2 – The effect of invasion by African Olive in Cumberland Plain Woodland on fuel load, structure and fire hazard was investigated. Additional studies included measurement of rates of decomposition of litter and comparison of overall flammability of invaded and native areas to better understand differences between pristine (uninvaded) sites and this novel fuel type.

Chapter 3 – Cootamundra Wattle is a native species that has invaded many ecosystems across Australia. In this chapter, the changes in fuel load, structure and fire hazard caused by this species are presented and related to how a species that evolved under the similar environmental conditions can affects the fuel and fire behaviour.

Chapter 4 – Data describing a range of common leaf traits, the mineral composition of leaves and their relationship to the four components of flammability for a range of weed and native species from woodlands and forests of eastern Australia are presented and interpreted.

Chapter 5 – Predictions of fire behaviour in a novel fuel type are presented in this chapter. Fire behaviour in the study area described in Chapter 2 is predicted using the BehavePlus Fire Modelling System and compared to predictions from existing models currently in use by Australian fire authorities.

Chapter 6 – The ways in which the findings from this study could be incorporated into existing frameworks and strategies dealing with weeds and their consequence for fire behaviour in the invaded areas are discussed. Suggestions for further studies relating to fire management in weed-invaded areas are also provided.

1.3. Weeds in Australia

The term 'alien species' was coined by Charles Darwin but its meaning has been updated many times to arrive at the concept that is accepted today. An alien species can be defined as any species located in an area as a consequence of human-mediated transport (Lockwood *et al.* 2007). Alien plants are also referred to as weeds, exotics, invaders, noxious plants and non-natives. The Australian Weeds Committee has defined 'weeds' as all plants that growing in unwanted places, damaging the economy, society and environment of the country (Australian Government Department of the Environment and Water Resources 2006). Relevant authorities in the state of NSW consider a weed to be any species that establishes and expands its range threatening ecosystems or habitats, or any species which can cause economic or environmental harm (Government of New South Wales 1993). Each state and territory in Australia has a similar definition but often follow separate legislation and guidelines to manage invasive species.

More than 15 years ago, the Federal, State and Territory ministers responsible for agriculture, forestry and the environment in Australia agreed to develop a National Weeds Strategy to reduce the impact of weeds on the sustainability of Australia's productive capacity and natural ecosystems (Sinden *et al.* 2004; Australian Weeds Committee 2005). In 1997, these actions culminated in the release of the National Weeds Strategy. In 1998, member states and territories agreed to organise a list of weed species (termed "weeds of national significance" (WONS)) according to an established set of criteria. In an attempt to organise knowledge about these species, the National Weeds Strategy Executive Committee maintains the list and basic data about each species which is regularly updated (Australian Weeds Committee 2005). The WONS list provides knowledge about exotic species harmful to the environment, however how these plants alter the environment and ecological processes still needs to be determined (Williams and Baruch 2000).

The number of plant species introduced to Australia since European settlement is estimated to be about 25 000 (Groves *et al*. 2002). State and Federal governments have classified over 370 plant taxa as noxious weeds, and it has been estimated that there are around 2700 non-native species registered as naturalised (a species that can form self-maintaining populations) (Groves *et al*. 2003, Groves *et al.* 2005). About 30% of these species represent a major threat to native plants including some endangered and endemic flora (Groves *et al.* 2003).

It is thought that since the time of European settlement the rate of introduction and spread of alien plants in Australia has increased linearly. However, this rate seems to be increasing exponentially in some areas in recent years (Adair and Groves 1998; Cook and Dias 2006). Kloot (1991) showed that between four and six new plant species are introduced to Australia per year and this number has been constant over the last 100 years. Since settlement, a number of species have become well established due to the inadvertent spread by primary industries as a wide range of species were used to develop agriculture and to make the country more competitive on the global market (Stone *et al.* 2008; Australian Weeds Committee 2014). Although many weed species were originally introduced for horticultural purposes (e.g. African Olive; see Chapter 2) or as ornamental plants (e.g. Cootamundra Wattle; see Chapter 3), it is important to remember that the environmental consequences of these plants were largely unknown or ignored (Stone *et al.* 2008). For example, Londsdale (1994) found that only 5% of non-native pastures plants introduced into Northern Australia between 1947 and 1985 have been useful to agriculture, while 13% have subsequently been listed as weeds. Of the useful species, 81% have become weeds on non-grazing land, with less than 1% proving to be beneficial to agriculture without any side effects.

Human technology has changed the world so much that the possibility for weeds to arrive in Australia has broadened considerably and weeds can now arrive via different vectors quickly and easily. Both Hulme (2009) and Mack and Lonsdale (2001) have called attention to the challenge of conservation in the light of species exchange in the modern world. In an era of "species globalisation" it is becoming more and more important to understand the routes, pathways and motivations for deliberate or unintentional introduction of weeds and to ensure that rigorous assessments are made to avoid biological invasion. A better understanding of the factors behind the success of invasive species in frequently disturbed environments and the role of invading species in altering ecosystems is a key step to produce knowledge to support management action (Flory and Clay 2010).

1.3.1 Weeds classes in NSW and ACT

The administration and control of weeds in NSW and the ACT is the responsibility of the Minister for Primary Industries under the *Noxious Weeds Act* 1993. There are five classes of weeds identified and described (Government of New South Wales 1993, p. 3-4):

"*Class 1 – Plants that pose a potentially serious threat to primary production or the environment and are not present in the State or are present only to a limited extent.*

Class 2 – Plants that pose a potentially serious threat to primary production or the environment of a region to which the order applies and are not present in the region or are present only to a limited extent.

Class 3 – Plants that pose a potentially serious threat to primary production or the environment of a region to which the order applies, are not widely distributed in the area and are likely to spread in the area or to another area.

Class 4 – Plants that pose a potentially serious threat to primary production, the environment or human health, are widely distributed in an area to which the order applies and are likely to spread in the area or to another area.

Class 5 – Plants that are likely, by their sale or the sale of their seeds or movement within the State or an area of the State, to spread in the State or outside the State."

Any noxious weed in NSW or the ACT that does not classify as a WONS and needs to be regulated by law will be included in one of the five classes above. The regulation of noxious weeds provides benefits to the community over and above the cost of implementing control programs (NSW Department of Primary Industries 2014).

1.4. Fire in Australia

Fire is a major environmental factor in Australian landscapes. The effects of fire are visible in nearly all vegetation types, differing from each other in terms of frequency of burning, fire intensity and fire season (Gill *et al.* 1981; Bradstock 2010). The frequency, intensity (rate of heat release during the burning process), seasonality (the time of year when the fire burns) and patchiness of the fire define what is referred to as the 'fire regime' of an area. The fire regime can alter vegetation structure and has the potential to influence plant invasion (Whelan 1995; Brooks *et al.* 2004; Mandle *et al.* 2011). The fire history of an area is the reconstruction of past and current fire regimes (Whelan 1995).

Fire has been part of the Australian landscape for at least 60.8 million years (Singh and Geissler 1985; He *et al.* 2011). There is still considerable discussion amongst the scientific community around the accuracy of this date due to its bias toward a small number of regions of the country and the simplicity of time resolution (Bradstock *et al.* 2002). At the beginning of the Tertiary period, the amount of precipitation was considerable (Scott 2000). Rainforest represented the majority of the vegetation and the climate and scarcity of eucalypts and other fire-promoting plants suggests that fires were usually isolated and did not affect large areas. The development of a drier climate and inconsistent patterns of rainfall towards the end of the Tertiary period meant that fires became more frequent and rainforest was replaced by fire-tolerant and arid-adapted open forest (Scott 2000; McLoughlin 2001). It is believed that before the arrival of humans in Australia, bushfires were frequent across the continent, particularly in the north where lightning was the principal source of ignition (Pyne 1990; Bradstock *et al.* 2002). After Aboriginal colonisation, approximately 40 000 years ago (Mulvaney and Kamminga 1999; Vigilante *et al.* 2009), it is thought that fire regimes changed with increases in fire frequency and changes in seasonality. At about the same time, there were major changes in the vegetation and there is still no agreement as to whether or not such changes were related to the prevalent weather conditions, anthropogenic activity or both (Gill *et al.* 1981; Singh and Geissler 1985; Scott 2000; Bradstock *et al.* 2002).

It is believed that the dominance of *Eucalyptus* in Australia since the Holocene may be an artifact of Aboriginal burning (Pyne 1990). The theory of "fire-stick farming" suggests that the arrival of Aboriginal populations in Australia caused changes in burning regimes and led to trophic-level shifts in ecosystems (Jones 1969). The fragmentation of woodlands and forests caused by fire and the expansion of grasslands, especially *Triodia* grasslands, created conditions for modification of the natural environment. Using fire as a tool, Aboriginal populations slowly altered environmental conditions such as nutrient availability and enhanced herbaceous plant productivity to create a mosaic in the landscape that allowed fire-prone communities to develop (Pyne 1990; Bird *et al.* 2008).

Characteristics of fire behaviour such as intensity, frequency and season of burning changed again after European settlement (Richards 1990; Ward *et al.* 2001; Jurskis *et al.* 2003; Watson and Wardell-Johnson 2004; Jurskis 2005; Burrows *et al.* 2009; Watson *et al.* 2009). Fire exclusion and fire suppression practices were used more frequently in settled areas, while planned burning was used in remote areas to reduce fire hazard or to improve grazing potential (Jurskis *et al.* 2003). Together with changes in fire regimes as a result of European colonisation, an important new factor was added to the pool of variables that can affect fire behaviour – the introduction of hundreds of new plant species that could potentially alter vegetated ecosystems and the quality and quantity of fuel. The alteration of the fuel structure and load caused by invasive species is investigated in Chapters 2 and 3. Changes in fuel chemistry and plant flammability are investigated in Chapter 4.

1.5. Development of modern Australian flora in a fire-prone continent

Characteristics that favour plant survival after bushfires in Australia may have evolved for reasons other than fire. According to Keeley *et al.* (2011), plant species are not 'fire adapted'. They are adapted to a particular fire regime, which, among other things, includes fire frequency, fire intensity and patterns of fuel consumption. For example, the ability to resprout may have evolved in response to grazing or drought and hard seed coats and woody fruits may be a response to low soil nutrients (Keeley and Zedler 1998; Schwilk and Ackerly 2001). The evolution of traits that improve the fitness of plant populations found in fire-prone landscapes has been the focus of fire research for a number of years and is a controversial field (Saura-Mas *et al.* 2010; He *et al.* 2011; Bowman *et al.* 2014).

The contemporary terrestrial flora of Australia is markedly different from that of other continents. Australia has an enormous number of species, genera and families that are endemic and many other taxa have Australia as their centre of diversity. The typical look of much of the vegetation in Australia is due to the dominance of *Eucalyptus* and *Acacia* in forests and woodlands that cover over 70% of the continent (Christophel 1989;

Hill 2004). The Australian flora has a distinct Gondwanan origin. Remnant rainforest ecosystems representing what was once a diverse and widespread flora in the early Tertiary period are now represented by plants confined to moist, sheltered habitats (Cowling and Lamont 1998; Pennington *et al.* 2009). Since the middle of the Tertiary period, the majority of plants in Australia diversified, especially in temperate and semi-arid environments (McLoughlin 2001). As an example, the flora of the Southwest Australian Floristic Region is primarily represented by angiosperms, especially woody families: Myrtaceae (1283 species/subspecies), Proteaceae (859), Fabaceae (540), Mimosaceae (503), Orchidaceae (374), Ericaceae (including Epacridaceae, 297), Asteraceae (280), Goodeniaceae (207), Cyperaceae (199) and Stylidiaceae (178) (Hopper and Gioia 2004). The importance of woody taxa is also evident in the ten largest genera. Listed in order of number of species/subspecies they include: *Acacia* (Mimosaceae; 502 species/infraspecies), *Eucalyptus* (Myrtaceae; 362), *Grevillea* (Proteaceae; 229), *Melaleuca* (Myrtaceae; 185), *Leucopogon* (Ericaceae; 165), *Verticordia* (Myrtaceae; 138), *Dryandra* (Proteaceae; 136) and *Hakea* (Proteaceae; 105) (Hopper and Gioia 2004). Of these, only the herbaceous Trigger plants (*Stylidium*, Stylidiaceae; 170 species) and the geophytic orchid *Caladenia* (Orchidaceae; 162 species) constitute non-woody plants (Hopper and Gioia 2004). Most of these species have some features that allow them to survive in low nutrient soil and to withstand seasonal water and heat stress and fire.

With an increase in aridity and the prevalence of fire towards the end of the Tertiary period (Kershaw *et al.* 2002), sclerophyllous plants became more common (Hill 2004). Many of their characteristic features promoted fire resistance, for example, trees having thick, insulating bark on the lower half of the trunk, resprouting after vegetative damage from dormant buds located on lignotubers, at the base of woody stems and along trunks or stems and stimulation of flowering and mass production of seeds after fire (Keeley *et al.* 2011). Fire can kill some plants but these species may have the ability to reproduce from seed protected in hard, woody cones (e.g. Cypress pine or *Callitris*) or fruits, such as follicles (e.g. *Hakea*, *Banksia*) and capsules (e.g. *Eucalyptus*, *Leptospermum*). Other species have a seed coat which can only be cracked by heat. Fast-growing acacias are often among the first woody plants to regenerate after fire (Floyd 1966; Christophel 1989; Hill 2004).

The tolerance of a plant to environmental processes is regulated by an array of functional traits and there is a feedback relationship between traits and processes (Scarff and Westoby 2006). Plant flammability relates to a set of traits that regulate and are regulated by the fire events in an ecosystem resulting in a multi-level complex feedback relationship (Gill and Moore 1996; Pausas *et al.* 2004; Gill and Zylstra 2005; Scarff and Westoby 2006; White and Zipperer 2010). These relationships will be investigated in Chapter 4.

1.6. Fuel types

The fire environment is broadly composed of weather, fuel and topography (Countryman *et al.* 1972). These three factors constantly interact (Agee 1997) and a change in any one of these factors will cause a change in fire behaviour (Whelan 1995). This section will therefore focus on what is known about the fuel component and how it can affect fire behaviour. Fuel is described as an irregular array of combustible elements with spaces that must be crossed by fire for new fuels to become available (Zylstra 2010). Sullivan *et al.* (2012) have defined fuel as a generic term used to describe any combustible material. Fuel can also be defined as any burnable living and dead vegetation that may be consumed in the passage of a fire (Whelan 1995; Cochrane and Ryan 2009).

Sullivan *et al.* (2012) noted that a fire is often described according to the predominant fuel type in which it is burning (e.g. grass fire, forest fire). Different classifications of fuel type have been described and used by land and fire management services around the world (Arroyo *et al.* 2008). However, describing fuel properties is usually very complex and it is therefore common practice to group vegetation types according to similar fire behaviour characteristics (Riaño *et al.* 2002). Knowledge about plant species alone is not enough for fire management, given that the same species could represent completely different fire behaviour because of different growth habits and/or fuel accumulation and decomposition in different environments (Anderson 1982; Andrews 1986). Merril and Alexander (1987, p. 24) defined fuel type as "*an identifiable association of fuel elements of distinctive species, form, size, arrangement, and continuity that will exhibit characteristic fire behaviour under defined burning conditions*". This definition takes into account the composition, structure and arrangement of fuel and the role that these elements have in fire behaviour. A map of the key fuel types of Australia according to the dominant fuel layer has recently been published (Sullivan *et al*. 2012). Three categories that represent the majority of Australian vegetation from a fuel perspective are represented and current fire behaviour models for each of the vegetation types – grassland, forest and shrubland – are summarised below.

Fuel is defined by its physical attributes such as load, depth, height, bulk density, particle size, and proportion of live and dead material (Gould *et al.* 2011). A qualitative and quantitative description of fuel is important for understanding fire behaviour and provide information for fire management activities (Sandberg *et al.* 2001, Cruz *et al.* 2010).

Methods that allow quick assessment of fuel characteristics with reliable consistency are of increasing interest to fire managers, ecologists, air quality managers and carbon modellers (Ottmar *et al.* 2003, Gould *et al.* 2011). Different countries have developed their own methods for describing fuels according to the vegetation types being managed. Here, the three main systems developed in Australia, Canada and the United States are briefly described and compared with requirements for variation or adjustment for use in other countries and vegetation types (see Section 1.10.).

1.6.1. Grassland

Grassland is defined as an area dominated by grasses (Family *Poaceae*) rather than large shrubs or trees (Watson 1990). Grasses have a worldwide distribution and are the most dominant and economically important family of flowering plants. They occur in virtually all major ecosystem types and cover about 30% of naturally vegetated areas worldwide (Watson 1990; Singh and Upadhayaya 2000).

According to Cheney and Sullivan (2008), grasslands in Australia can be divided into five groups: tropical, tussock, hummock, improved pastures and croplands. Other types of grasslands (e.g. alpine feldmarks) and tussock grasslands intermingled with herbfields, sedges and rushes are small in extent (around 64 000 km^2) and fire behaviour in these vegetation types are similar to fire in more common types of grassland (Annon. 2006). In addition, Cheney *et al.* (1998) described three conditions for this general fuel type: (1) undisturbed and/or very lightly grazed natural grasslands or improved pasture or unharvested crops, generally more than 50 cm tall, (2) grazed or mown pasture, generally less than 10 cm tall, and (3) very heavily grazed pasture, generally less than 3 cm tall with scattered patches of bare ground.

Abiotic factors such as rainfall, wind, temperature and photoperiod can influence the phenological cycle of grasses (Veenendaal *et al.* 1996a; b). However, grasses have the ability to adapt their life cycle to variations in the hydrological cycle due to seasonality and fluctuation in average annual rainfall (Veenendaal *et al.* 1996a; b; Munhoz and Felfili 2005). If conditions are favourable during the growing period then biomass will be accumulated. Once grasses have stopped growing or their growth slows down, combustibility then depends on the rate of curing (Sullivan *et al.* 2012).

1.6.2. Forest

A 'forest' is defined as an area of more than 0.5–1.0 ha with a minimum tree crown cover of 10–30%, with 'tree' defined as a plant with the capability of growing to be more than 2 to 5 m tall (UNFCCC 2002). The most recent report shows that Australia has 147.4 million ha of native forest mostly dominated by eucalypts and acacias and around 2.0 million ha of plantations (ABARE 2011). Together, forests cover about 19% of the continent and represent about 4% of the world's forests. The current distribution of tree species in Australia is the outcome of the interaction of several factors with one of the main ones being the ability to survive periodic bushfires (Boland *et al.* 2006).

Fire-prone Australian forests are represented by three main vegetation types: dry sclerophyll forest, wet sclerophyll forest and woodlands (Boland *et al.* 2006). Dry sclerophyll forests have low rates of primary production and low rates of breakdown or decomposition of highly flammable leaves. The overstorey canopy as well as many plants in the understorey shrub layer and the eucalypt bark are also highly combustible (Boland *et al.* 2006). Dry sclerophyll forests usually replace wet sclerophyll forest in areas where rainfall is less than 600 mm. Dry forests are also found in areas with higher rainfall but only where soil has low nutrient availability or is too shallow to retain adequate moisture (Turnbull 1997; Boland *et al.* 2006). Shrubs or grasses form the understorey of both forest types depending on fire regime, canopy openness, soil and rainfall (Boland *et al.* 2006). Woodlands typically have an open canopy, composed of different species of *Eucalyptus* and have an understorey dominated by grasses and forbs and occasionally with shrubs (Benson and Howell 2002; Hill *et al.* 2005).

Gould *et al.* (2007a) divided forest fuels into two categories: the 'surface stratum' encompassing litter, near-surface and elevated fuel layers and the 'canopy stratum' encompassing the intermediate and overstorey canopy layers (each layer is detailed in Section 2.2.3). Litter fuels consist of fallen leaves, bark and twigs that are usually layered horizontally and represent the majority of the fuel consumed by fire on a weight basis (Sullivan *et al.* 2012). These fuels are responsible for the greatest proportion of the energy released by fire during a forest burn (Whelan 1995; Gould *et al.* 2011). Near surface-fuels correspond to grasses, shrubs, creepers and collapsed understorey and suspended material such as bark and twigs shed by overstorey trees (Gould *et al.* 2011). Elevated fuel includes shrubs and young trees that compose the midstorey strata. The density and height of elevated fuel dictates some characteristics of fire behaviour, especially flame height. Bark fuel is directly associated with the trees that compose the intermediate and overstorey strata and is a key factor in ember production and spotting (Sullivan *et al.* 2012). Eucalypt bark fuel can have an important role in fire spread by contributing to spot fires away from the original fire (Sullivan *et al.* 2012).

1.6.3. Shrubland

The Mediterranean climate is situated between the parallels 30 and 40 North and South and can be found in five regions around the world: the Mediterranean Basin, California, central Chile, South Africa and south-western and southern Australia (Di Castri *et al.* 1981). These regions are mostly dominated by evergreen shrublands and heathlands (Arroyo and Maranon 1990). They are characterised by the presence of evergreen woody sclerophyllous shrubs with an occasional overstorey of small trees and a herbaceous understorey (Di Castri *et al.* 1981; Arroyo and Maranon 1990; Grooves 1991; Rambal 2001).

In Australia, shrublands occurs in a variety of climates and develop a vertically uniform but spatially discontinuous fuel complex (Parsons 1994). Most studies of fire behaviour in Australian shrubland have been done in mallee-heath (Bradstock and Gill 1993; McCaw 1997; Cruz *et al.* 2010). This vegetation type occupies approximately 270 000 km^2, it is composed mainly of multi-stemmed eucalypts and is found in areas with less than 300–350 mm annual rainfall (Cruz *et al.* 2010; Sullivan *et al.* 2012). The vegetation structure of shrubland promotes fire behaviour where a small change in fire spread can lead to large changes in fire behaviour (Catchpole 2002; Cruz *et al.* 2010). Dead fine fuel from tussock and hummock grasses can carry fire through the near surface layer (Bradstock *et al*. 2002; Sullivan *et al.* 2012). In non-mallee-heath shrublands, the elevated fuel layer is composed mainly of species from the genera *Banksia, Leptospermum, Hakea* and *Dryandra*, and there may be a large amount of suspended bark hanging from the stems. This layer is important for fire behaviour of Australian shrublands as it contains a substantial proportion of dead fine fuel and live fuel that contains volatile terpene compounds capable of forming a highly flammable 'ladder' for a surface fire to move into the canopy (Cruz *et al.* 2010).

1.7. Fire behaviour

Fire behaviour is defined as the way in which a fire develops in response to factors in the environment related to fuel, weather and topography (Whelan 1995). Fire intensity, rate of spread and flame dimensions are the principal aspects related to fire behaviour (more details are provided in Chapter 5) and each of these aspects are influenced by the others making fires a very complex and dynamic phenomena (Whelan 1995). For example, flame height during fire and post-fire scorch height can be related to the fire intensity for some vegetation communities such as eucalypt forest in Australia (Luke and McArthur 1978) and pine forest in North America (Wagner 1973).

Fuel type, quantity, arrangement and moisture content are all attributes that can affect fire behaviour. The quantity of fuel is defined as the amount of surface fuel on or near the ground and the amount of bark and elevated fuel above the ground (including shrubs and suspended materials) and is usually measured in small areas and scaled up to t ha^{-1} (Whelan 1995). The arrangement and distribution of fuel is also important for determining fire behaviour. A continuous loose fuel layer allows fire to move faster through it than in a compacted or discontinuous fuel bed (Whelan 1995). Most of the fire models in Australia currently underestimate fuel complexity and arrangement (Gould *et al.* 2007a; Cruz *et al.* (2010). However, Gould *et al.* (2007a) proposed that the fuel complex should be looked at as being two separate layers; a lower stratum encompassing litter, near-surface and elevated fuel layers, and the canopy stratum encompassing the intermediate and overstorey canopy layers. This change in perspective has provided a more accurate fire behaviour model in comparison to the classic McArthur model when predictions have been applied to real fires.

The dead fine fuel and air moisture also have an important influence on each other and on fire behaviour overall. Luke and McArthur (1978) defined fine fuel as all vegetation, living or dead, measuring 6 mm in diameter or less. Fine fuel is responsible for maintaining the flame front of a fire. In contrast, heavy fuels, such as logs, requires combustion of fine fuel (Whelan 1995). The moisture content of fuel is the amount of moisture expressed as the percentage of its dry weight (Sullivan *et al.* 2012). The moisture content of fuel affects how intensely a fire will burn. For example, for dead fine fuels, the moisture content is related to how dry the season has been and in the absence of rain, moisture content is governed by the dryness of the surrounding air or the relative humidity (Whelan 1995; Cheney *et al.* 1998; Catchpole 2002b; Matthews 2010; Matthews *et al.* 2010).

1.7.1. Rate of spread and fire intensity

Rate of spread describes how quickly the edge of a fire is moving, usually it is given in m h^{-1} or m s^{-1} (Whelan 1995). Rate of spread is the primary output of most fire behaviour models and there is an extensive literature of models used to predict fire spreading through different vegetation types (Sullivan 2009a). Peet (1965) and McArthur (1962, 1967) defined the rate of spread of fire (ROS) as being directly proportional to the fine fuel load (<6 mm diameter) consumed (w) and expressed as a linear relation where a is their defined constant (these relationships are explained in depth in Chapter 5):

$$\text{ROS} = aw \qquad [1.1]$$

McArthur (1962, 1967) proposed that the amount of fuel available on the forest floor is one of the most important variables affecting rate of spread. However, this relationship is weak and only exists during small fires in very specific fuel types (Gould *et al.* 2013). It has been shown that factors other than fuel load are more influential. Rate of spread varies around the fire and is faster at the downwind edge known as head of the fire or fire front. It also depends on weather (Fons 1946; Cheney *et al.* 1993; Gill *et al.* 1996; Cheney and Sullivan 2008), fuel attributes other than load (Fang and Steward 1969; Burrows *et al.* 1991; McAlpine 1995; Cheney *et al.* 1998; Burrows 1999; Gould *et al.* 2007a) and topography (Whelan 1995).

Fire intensity (kW m^{-1}) is the energy released per unit length of fire front. Calculation of fire intensity includes the amount of fuel being consumed and the rate of spread (Luke and McArthur 1978) using the equation developed by Byram (1959):

$$\text{I} = \text{H w ROS} \qquad [1.2]$$

where H is the heat yield of the fuel, w is the load of the consumed fuel (kg m^{-2}) and ROS is the rate of spread of the head fire (m s^{-1}).

Fire intensity and total energy released per unit area are two of the most important variables estimated from a fire. The tolerance of heat in living cells/organisms is associated with these two variables and they also correlate with other factors such as scorch height of vegetation (Rothermel and Deeming 1980; Whelan 1995).

1.7.2. Flame dimension and definitions

Parameters such flame length, height, depth and angle define the flame dimension. Whelan (1995) and Keeley (2009) highlighted the difficulties of measuring components of fire intensity in the field and showed that the flame dimensions can be related to many aspects of fire intensity.

Flame length is defined as the length of a flame measured along its axis at the fire front; the distance between the flame height tip and the midpoint of the flame depth at the ground surface (Figure 1.1). Flame length is often used as an approximate indicator of fire front intensity. Flame height is the average maximum vertical extension of flames at the fire front. Flame depth is the width of the zone within which continuous flaming occurs behind the edge of a fire front. Flame angle is the angle formed between the flame at the fire front and the ground surface, expressed in degrees (Merrill and Alexander 1987), but all flame dimensions may be expressed as the angle between the flames and vertical (Figure 1.1).

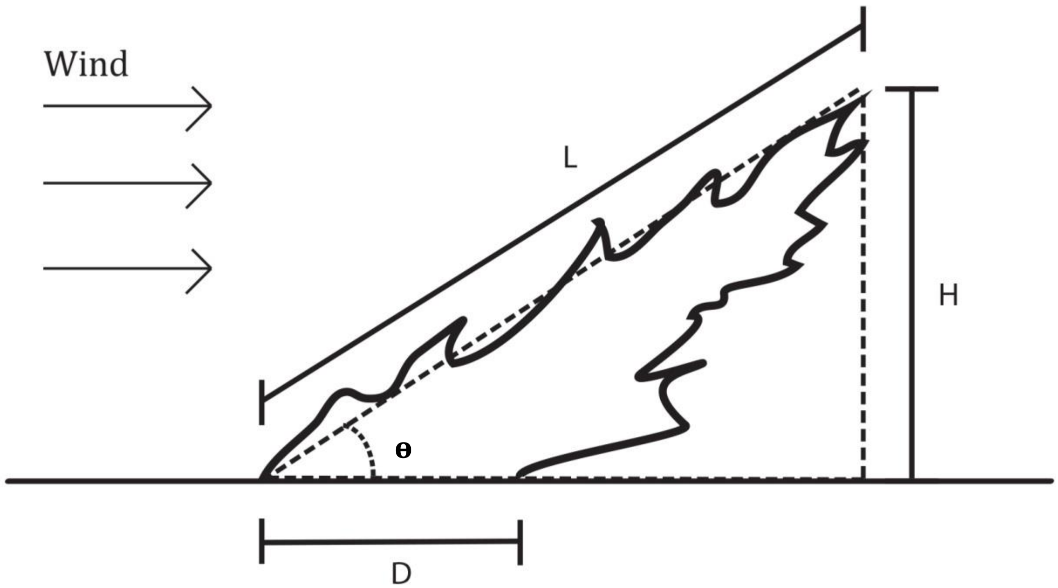

Figure 1. 1. Flame length (L), height (H), depth (D) and angle (ɵ) (adapted from Cheney and Sullivan 2008).

1.8. Influence of native Australian fuels on fire behaviour

1.8.1. Grassland fire behaviour

Grasses are classified as fine fuels and are characterised by having a high surface area-to-volume-ratio, high thermal conductivity, low density, vertical orientation and continuous distribution (Sullivan *et al.* 2012). Grassfires burn relatively quickly with a rapid rate of combustion, a high rate of spread and with a characteristically rapid response to wind changes (Cheney and Sullivan 2008). The average flame height of fires in savannah grasslands are between 0.8–2.8 m (Frost and Robertson 1987) and average residence times (average time with visible flames) of flames are 5–15 s (Cheney and Sullivan 2008). Fire intensities for savannah fires in Australia can vary from 100–18 000 kW m^{-1} depending on the wind speed and fuel moisture (Griffin and Friedel 1984; Morgan 1999). Similar values for fire intensity have been recorded for fires in savannahs in Africa and Brazil (Frost and Robertson 1987; Miranda *et al.* 1996).

The rate of spread of grassfires under extreme weather conditions can vary from 15 km h^{-1} to more than 30 km h^{-1} (Noble 1991, Cheney *et al.* 1998, Sullivan 2010), and is influenced by the curing state of the fuel. Morgan (1999), working in tussock grassland vegetation of western Victoria, found a positive correlation between fire intensity and rate of forward spread, but no correlation between rate of spread and fuel load as predicted by McArthur (1967). In addition, a continuous layer of grass that is less than 50% cured does not sustain a fire and results in a patchy burn (Cheney *et al.* 1993; Cheney *et al.* 1998; Cheney and Sullivan 2008). If the rate of spread for grassfire is modeled according to moisture content and wind speed, the minimum threshold for a fire to be maintained is approximately 20% fuel moisture content and a wind speed of 10 km h^{-1} (Cheney *et al.* 1998).

1.8.2. Forest fire behaviour

In forests, the near surface fuel is the principal fuel layer responsible for the rate of spread of fire (Gould *et al.* 2007a). There may be an increase in rate of spread in older fuels, but in most cases, surface fuel load is the only attribute that is available to quantify this effect. In older fuels, it is thought that other fuel layers such as near-surface and elevated fuels are more important in determining rate of spread than surface fuel alone (Gould *et al.* 2007a). In addition, bark fuel plays an important role in rate of spread due to spotting potential. The rate of spread of fire in forest can be directly related to wind speed measured at 5 m in the forest above a threshold wind speed of about 1 m s^{-1} (Gould *et al.* 2007a). Although rate of spread is weakly related to fuel load it can be directly related to attributes of the surface fuel bed and the understorey fuel layer.

In forest fires, fire intensity is influenced by fuel moisture content. The relationship between quantity of available fuel and moisture level and the rate of combustion are determinant aspects to the fire intensity (Sneeuwjagt and Peet 1985). Increased fire intensities are often a combination of high fuel levels and low fuel moisture (Cheney 1981; Sneeuwjagt and Peet 1985; Burrows 1994; Smith *et al.* 2004). An increase in fire intensity is indicated by an increase in flame height and heat output (Whelan 1995; Smith *et al.* 2004). Heat yield (H) for eucalypt forests can be as high as 18 600 kJ kg^{-1} (Byram 1959) but fire intensities can range from low (less than 350 kW m^{-1}) to high (350–3500 kW m^{-1}) to very high (3500–5000 kW m^{-1}) to extreme (greater than 5000 kW m^{-1}; (Cheney 1981; Gill and Catling 2002). The combustion of coarse woody debris contributes mainly to total energy output and rate of heat release (Byram 1959; Rothermel and Deeming 1980). The ability to accurately predict consumption of coarse woody fuel is important because this fuel is readily available during a fire and information for Australian southern eucalypt forest fires is still limited (Hollis *et al.* 2010).

1.8.3. Shrubland fire behaviour

Fire behaviour in shrubland or mallee-heath is characterised by rapid changes due to a strong dependency on weather conditions which can change the ability of fire to overcome fuel gaps and move between fuel strata (Sullivan *et al.* 2012). Fuel moisture of dead fine fuels has been described as the principal variable influencing fire continuity in mallee-heath (Cruz *et al.* 2010). Along with moisture content of dead fine fuel, wind speed and fuel cover are very important variables for determining fire spread in mallee-heath. Strong winds are necessary to tilt the flames forward, pre-heating and igniting fuel (Sullivan *et al.* 2012).

The horizontal and vertical discontinuities that characterise the mallee-heath fuel complex prompt unexpected changes in fire spread. In low intensity fires, the flames travel on the litter and near-surface fuels and fire spread can slow or cease due absence/discontinuity of fuel. If enough fuel is present and the wind conditions are suitable, fire intensity increases and the flame front gradually evolves and is determined by elevated fuels and the overstorey canopy (Cruz *et al.* 2010; Peterson *et al.* 2011).

Despite wide variation in structure, floristic composition and environmental conditions amongst mallee-heath communities, fire intensities can vary from 9–35 MW m^{-1} (Keith *et al*. 2002). The forward rate of spread can vary from 0.33–2.08 m s^{-1} but the minimum threshold for a crown fire to occur in this type of fuel is a forward rate of spread greater than 0.4 m s^{-1} and a fire intensity greater than 8.5 MW m^{-1} (McCaw 1997). Different approaches to modelling fire spread in mallee-heath vegetation have been developed although none are broad enough to be applicable for general use (McCaw 1997; Cruz *et al.* 2010; Sullivan *et al.* 2012).

1.8.4. Influence of invasive plants on fire behaviour

Plant invasion and fire can significantly change ecosystems (Fisher *et al.* 2009). Fire influences and is influenced by plant community composition and structure resulting in a complex relationship between fire behaviour and weeds (Mandle *et al*. 2011). Invasion by exotic grasses has been shown to cause an increase in fire frequency (D'Antonio and Vitousek 1992; Balch *et al.* 2013). For example, the density of Pompom Weed (*Campuloclinium macrocephalum*), an invasive forb in grasslands in South Africa, increased with increasing fire frequency (Goodall *et al*. 2011). The long-term interaction of fire with weed invasion can change ecosystems from savanna to shrubland (Walther *et al*. 2009), grassy woodland to shrub-dominated woodland (Watson *et al*. 2009) and shrubland to woodland (Keeley 2011).

Differences in growth rate, architecture and ecophysiological characteristics among invasive and native vegetation can alter fire regimes and produce significant changes in the balance of carbon, nutrient levels and the water cycle (D'Antonio and Vitousek 1992; Pyšek *et al.* 2009). As such, many studies have reported changes in the biogeochemical cycles at a range of scales due to the presence of invasive plants (D'Antonio and Vitousek 1992; Hobbs and Huenneke 1992; Rew and Johnson 2010; Allen *et al.* 2011; Mandle *et al.* 2011).

There are a number of characteristics of weeds that can influence fire behaviour or modify fire frequency in weed-invaded areas. These include increased biomass accumulation through high primary productivity and shedding of leaves and branches that increase the fuel load (Brooks *et al*. 2004; Balch *et al.* 2013), increased flammability due to oils that facilitate ignition of plant material (Allen 2008), and structural changes in the vertical distribution of fuels (Pauchard *et al*. 2008). The impact of Lantana (*Lantana* *camara*) on fire behaviour in an Australian rainforest has been attributed to two primary mechanisms by which this species can alter the incidence of fire (Berry *et al*. 2011). The first mechanism is related to the introduction of a flammable material to a generally non-flammable ecosystem. The second mechanism is a change in fire occurrence by increasing the availability of fuel. For this species, alteration of fire behaviour is more likely to be due to a change in the fuel bed (facilitating fire intensity and spread) instead of by increasing biomass ignitability. Conversely, there may be a decrease in fire risk due to low surface area-to-volume ratio of weed biomass resulting in greater moisture retention (Grace 1998), fuel compression that leads to suppression of fire due to lack of oxygen (Van Wilgen and Richardson 1985), and alteration of the understorey layer due to shading caused by invasive trees (Brooks *et al*. 2004). Mandle *et al.* (2011) recently reviewed the major studies involving woody weeds and alteration of fire regimes. Of the 16 woody species identified in this study, eight species were found to increase fire frequency or fire intensity; five species decreased aspects of the fire regime and three species had mixed effects.

The effects of woody weeds on fuel load and structure, fire regimes and intensity remains poorly understood, particularly in an Australian context (Mandle *et al.* 2011). The introduction of weeds into an ecosystem requires new models to be developed or, more realistically, existing fire behaviour models to be adapted to incorporate different fuel types. Several studies have quantified fire behaviour in weed-invaded areas but most of these have been in grassland (Floyd 1966; D'Antonio and Vitousek 1992; Briese 1996; Fine 2002; Brooks *et al.* 2004; Stohlgren and Schnase 2006; Pauchard *et al.* 2008; Burrows *et al.* 2009; Cohn *et al.* 2011). Although invasion by herbaceous species and the mechanisms by which they change fire behaviour are beginning to be understood, the same is not true for woody weed species. In this thesis, aspects of change in fire behaviour due to woody weed invasion in forest ecosystems in south eastern Australia are investigated.

1.9. Modelling fire behaviour

The term "model" is used in numerous ways in predictions of fire behaviour creating confusion between fire managers, the scientific community and the general public (Harrington 2005). Hence, it is important to first clarify these different usages. According to Harrington (2005) some characterisations of forest fuels have been called *fuel models*. These "models" are profiles or sets of information about surface fuels that provide inputs to mathematical models of fire spread. Fuel models form a subset of approaches to characterising fuels and for the purpose of forest fire and fuels management simulation, the term *fuels characterization* is becoming the dominant terminology (Graham *et al.* 2004; Harrington 2005). *Mathematical models* were developed to predict fire spread in studies such as Rothermel (1972) and Van Wagner (1977). The term "model" can also be used for fire behaviour models. In this case it is used to refer to computer simulations based upon mathematical models that predict spread and intensity and show an interpretable graphic form (Harrington 2005). In this thesis, both 'fuel models' and 'mathematical models' will be used in fire behaviour predictions.

Measuring and modelling fire behaviour began in the early 1930s and today there is a continuum of approaches ranging from physical and quasi-physical models and empirical and quasi-empirical models to simulation and mathematical analogous models to measure and predict fires (Sullivan 2009b). Sullivan (2009a; b) reviewed the most important models in this continuum and concluded that a combination of the best attributes of the various approaches employed will lead to more usable models and prediction systems.

1.9.1. Development of fire behaviour and danger models in Australia

A fire danger rating system is defined as a managing system that incorporates the aspects of particular fire danger factors into qualitative numerical indices of current protection needs (Chandler *et al.* 1983,). Although fire behaviour is an important part of fire danger systems, the classification of fire danger takes in account other factors such as potential for ignition, fire spread and damage.

The most widely used prediction systems in eastern Australia are the McArthur Forest Fire Danger Rating System and the McArthur Grassland Fire Danger Rating System, both of which were developed by Alan McArthur in the 1960s (McArthur 1966, 1967). In Western Australia, fire behaviour is predicted using estimates of fuel detailed in Forest Fire Behaviour Tables (Sneeuwjagt and Peet 1985).

Luke and McArthur (1978) divided fuel type into two categories – grass and forest – and determined the rate of spread for representative samples of each fuel type. During the development of the Forest Fire Danger Meter, fire behaviour was based on single fires burning under commercial eucalypt forests (Luke and McArthur 1978; Gould *et al.* 2011). The metrics (meters) for estimating fire danger in grassland and forest vegetation types allowed the prediction of the expected rate of spread and the difficulty of containment over a large area (McArthur 1966, 1967; Cheney *et al.* 1998; Cheney and Sullivan 2008).

Most of the fire models and guides for fire spread in eucalypt forests were developed by correlating fire behaviour from small experimental fires and opportunistic observation of bushfires when fuel and weather parameters could be collected (McArthur 1962; McArthur 1973; Gould *et al.* 2007a). These models use a directly proportional relationship between rate of fire spread and fuel load. As a result, they predict that a 50% reduction in fuel load will halve the rate of spread but will reduce the fireline intensity fourfold (Cheney and Gould 1996; Fernandes and Botelho 2003). More recent studies (e.g. Gould *et al.* 2007b; McCaw *et al.* 2008; Zylstra 2011) propose alternative approaches such as fuel structure rather than fuel load to evaluate fire danger.

Burrows (1994) suggested that the models created by McArthur can under predict fire behaviour when there are severe weather conditions. Proof of this was provided by McCaw *et al.* (2008) when predictions from the Forest Fire Danger Meter, the Forest Fire Behaviour Tables, the fire spread model of Burrows (1999) and data acquired from Project Vesta (Gould *et al*. 2007a) were compared. Fires were shown to spread two to three times faster than predicted by the Forest Fire Danger Meter and Forest Fire Behaviour Tables models, and up to five times faster than predicted by the spread model of Burrows (1999) under severe weather conditions.

Recently, Gould *et al.* (2007a) documented the fuel characteristics, fire behaviour and rate of spread for dry eucalypt forest and introduced a new concept of how the fuel complex determines fire behaviour. These authors report that the increase in rate of spread of fire with increase in fuel load as demonstrated by McArthur (1967) in fuels of different ages might not be exclusively related to fuel load. Instead, a range of structural factors such as height, composition, continuity and greenness changes as fuel re-accumulates after fire.

The model proposed for Project Vesta (Gould *et al.* 2007a; Cheney *et al.* 2012) established relationships between fire spread and fuel of different age and indentified fuel characteristics which could be correlated with forward spread. The fuel structure, fuel moisture, fire behaviour and wind speed were combined in the model to generate more accurate predictions. Cruz *et al*. (2010) working in mallee-heath shrubland identified many distinct characteristics related to rate of spread and fire behaviour. Similar characteristics in other vegetation types such as woodlands and weed-invaded areas need to be better described and modelled.

1.9.2. Describing fuels in Australia – hazard scores

For many years, fire authorities, fire scientists and land managers have been developing systems to describe fuels and predict fire behaviour in Australian vegetation. Early attempts involved meters created by McArthur (1962; 1967) and (Peet 1965). Recent advances to assess factors affecting fire behaviour and suppression difficulty have used new approaches to assess fuel (Gould *et al*. 2011) and these are described below.

Visual fuel hazard rating systems have gained attention from the scientific community during the last 15 years (Cheney *et al.* 1992; Wilson 1992; Tolhurst *et al.* 1996; McCarthy *et al.* 1999; Gould *et al.* 2007b; Hines *et al.* 2010). A fuel hazard scoring approach assesses different fuel layers by evaluating percentage cover and the fuel hazard in each following the concepts of Cheney *et al*. (1992), Wilson (1992) or Tolhurst *et al.* (1996). Gould *et al.* (2007a) described five fuel layers for most Australian forests: (1) overstorey tree canopy layer, (2) intermediate tree and canopy layer, (3) elevated fuel layer, (4) near-surface fuel layer, and (5) surface fuel layer. These layers can be broadly identified by height and by changes in the bulk density, continuity and amount of live material (Gould *et al*. 2011). The percent cover score (PCS) and fuel hazard score (FHS), represent a subjective assessment of the flammability of each layer and are described in depth in Section 2.2.3.

Gould *et al.* (2011) advocated that systems that stratify forest fuels into different layers and visually score their attributes are robust, reliable and easy to use. In their opinion, the hazard score of different fuel strata can exhibit patterns of change with time after fire that reflects fuel accumulation. When combined with weather variables, visual hazard scoring systems can be used to predict rate of spread, fire intensity and other fire behaviour techniques (Gould *et al.* 2011). Although hazard score classification systems are beginning to be used by the scientific community and fire managers in general, Watson *et al.* (2012) showed that the subjective nature of the measurements can lead to inconsistency in the results and affect firefighting safety and effectiveness.

1.9.3. Describing fuels in Canada – characteristic fuel types

In Canada, the Canadian Forest Fire Danger Rating System (CFFDRS) is widely used (Van Nest and Alexander 1999). The CFFDRS has two major subsystems. One has been used throughout Canada since the 1970s and provides numerical ratings of relative fire potential for standard fuel types on level terrain based only on weather observations. This subsystem is called The Canadian Forest Fire Weather Index (FWI). The second subsystem is the Canadian Forest Fire Behaviour Prediction (FBP). This subsystem detects variability in fire behaviour amongst fuel types for a given slope incline in quantitative and descriptive terms based on certain FWI system components as inputs (Van Nest and Alexander 1999).

The FBP system predicts the rate of spread, fuel consumption and fire intensity of fires using some inputs from the FWI system. The organisation of fuels types in the FBP system divide them into two major groups based on readily available inputs. At present, the FBP system can be applied to 16 different fuel types and aims to provide inputs for the prediction of fire behaviour (De Groot 1993; Alexander *et al.* 1996; Hirsch 1996). The fuel types in the FBP are described in a qualitatively way, using terms that describes stand structure and composition, surface and ladder fuels and the forest cover and organic layer (FCSSDD 1992). Fire managers are required to select the fuel type that best suits the particular situation they are considering. Together with the inputs from the FWI system, the FBP system gives quantitative estimative of head fire spread rate, fire intensity, type of fire, and elliptical fire area, perimeter, and perimeter growth rate. These data can be easily accessed in the field from tables or computer software although a simplified method of assessment is provided in a field guide (FCSSDD 1992; Van Nest and Alexander 1999).

1.9.4. Describing fuels in the United States – fuel models and biomass photo series

Several methodologies have been developed in the United States to distinguish surface fuels and to provide data for mathematical models that suits the dominant forest types (Harrington 2005). Burgan and Rothermel (1984) demonstrated that to build a fuel model it is necessary to describe the vegetation as a fuel complex rather than precisely measuring its biomass, although both are related. The fuel complex is defined as the load and arrangement of fuel in an ecosystem and is conventionally divided into three layers: ground fuels, surface fuels and canopy (Clar and Chatten 1954; Harrington 2005).

Brown (1974) and Brown *et al*. (1982) described traditional inventory methodologies that estimate fuel load by weight per area unit in five components of the surface fuel layer: litter, herbaceous cover, downed woody debris, shrubs and tree regeneration. A major part of fire behaviour prediction systems used in the United States need fuel inputs to be represented by fuel models. Characteristics of fuel components can be described using many different variables, including heat content, mineral content and density, although the most common variable used for different applications is fuel loading or biomass per area unit (Pyne 1984; Keane 2013). The quantification of some of these components is seasonally sensitive and because of fuel heterogeneity, this kind of data collection can be difficult and time consuming (Harrington 2005).

In the United States, data collection in complex fuel types requires the use of different methods best suited to the fuel component (i.e. grass, shrubs, litter and slash). For example, the collection of data for surface fuel is difficult and time consuming and has led to the development of standardised fuel models to save resources (Anderson 1982; Harrington 2005). These models were developed by Frank Albini in 1976 and were originally called Northern Forest Fire Laboratory fuel models. However, after the development of a visual key and the widespread use amongst fire managers and authorities they started being recognised as "Anderson's 13 fuel models". The visual key involved a series of photos used as a guide for estimating surface fuel loads. Each photo series represents a particular forest type and a visual comparison is used to approximate the fuel load present in the forest to estimate the inputs that will be used to predict fire behaviour (Harrington 2005).

Although Anderson's 13 fuel models have been widely used by land managers for many years, they only consider the surface fuel layer and do not describe ladder or canopy fuels (Harrington 2005). Sandberg *et al.* (2001) found that the Anderson's 13 fuel model did not correlate with actual fuel loadings, vegetation cover, remote-sensing signatures or modelled ecosystem dynamics. Many modellers have used the Anderson's 13 fuel model to infer variables that could not be correctly predicted thus generating large approximation errors. However, a growing need for a more robust way to assign fuel loading has started efforts to create a more comprehensive fuelbed classification system. Within the last 10 years, a fuel characteristic classification (FCC) system that potentially provides a better fuel characterisation than Anderson's 13 fuel models has been developed (Ottmar *et al*. 2003). The FCC system is based on fuel bed descriptions to predict the kind of fuel most likely to be present and the quality and relative abundance of the fuel (Sandberg *et al*. 2001).

The FCC design permits users to select fuelbed descriptions or to modify existing ones to customise a new fuel bed. When users have their qualitative and quantitative fuelbed data, the FCC generates quantitative fuel characteristics such as physical, chemical and structural properties and probable fire parameters specific to the fuelbed in question. The FCC has not yet been incorporated into current simulation programs and currently all FCC classes are assigned to one of Anderson's 13 fuel models (Harrington 2005).

1.10. Predicting fire behaviour – BehavePlus and Rothermel models

Fire and land managers make use of a range of computer models to enhance their capacity for fire behaviour predictions. Fire behaviour simulations are generated by these models and different models are used to address different problems. Predictions of fire behaviour for new fuel types such as areas invaded by woody weeds can be done either through building a new empirical model or using descriptive characteristics of the new fuel type as inputs to pre-existing physical models. Prediction of fire behaviour in a novel fuel type generally rules out the use of empirical models due to a lack of suitable field data and the time and effort required collecting this data. Of the physical models currently in use (Weber 1991; Forbes 1997; Grishin 1997; Linn and Harlow 1997; Dupuy and Larini 2000; Morvan and Larini 2001; Asensio and Ferragut 2002; Séro-Guillaume and Margerit 2002; Zhou *et al.* 2005; Mell *et al.* 2007), only the BehavePlus model is able to deal with novel fuel types to offer outputs comparable with the models currently used to predict fire behaviour in Australian fuels across different fuel types.

Rothermel and Deeming (1972) first developed a mathematical model to estimate the rate at which a fire can spread through a homogenous fuel bed containing fuel particles of mixed size. The theoretical basis for the fire spread model was first developed by Frandsen (1971, 1973) with notable improvements by Rothermel (1972) and Albini (1976). The resulting model has been used to predict certain aspects of fire behaviour with a fair degree of accuracy based on correlations between scorch height, fireline intensity, heat per area unit, flame length and rate of spread (Burgan and Rothermel 1984). Consequently, this model is still used in fire behavior software such as BehavePlus and other prediction systems to estimate the rate of spread of fire through complex fuels (Andrews 2013).

The BehavePlus system is a series of interactive fire behaviour computer programs for estimating wildland fire potential under different fuels, weather and topographic situations (Burgan and Rothermel 1984). All mathematical models used by the BehavePlus model requires a full description of the fuel properties as inputs to calculations of fire danger indices or fire behavior potential and are an important tool to manage fires for a vast range of activities and objectives (Sullivan 2009a; b).

1.11. Aims

This study aimed to investigate the interaction of fire and invasive plant species in forests and woodlands of eastern Australia by comparing biotic and abiotic features of pristine and invaded ecosystems. The alteration of fuel load and the vertical and horizontal distribution of fuel biomass in woodlands invaded by woody weeds were assessed in two case studies. The first case study considered invasion of African Olive (*Olea europaea* ssp. *cuspidata*) in woodlands in New South Wales (NSW). Data was compiled to build a model to predict fire behaviour in this novel fuel type. The second case study aimed to investigate differences in fuel structure and load among sites invaded by Cootamundra Wattle (*Acacia baileyana*), a native invasive species, and native woodland in the Australian Capital Territory (ACT). In both case studies, the flammability of the woody weed was compared to vegetation from the surrounding native woodland to enhance understanding of fire behaviour in different vegetation types and to highlight potential differences in flammability between native and non-native weeds. Following this same line of investigation, the relationships among flammability, leaf morphology and plant chemistry of a range of invasive and native woody plants were explored.

Using the data acquired from field and laboratory analyses we aimed to build and test a fuel model to predict the fire behaviour in areas invaded by African Olive. Fire predictions outputs were compared with outputs from models currently in use by Australian fire authorities. Such comparisons allowed assessment of the main differences among fire behaviour prediction models.

Overall, the aim of this research was to provide high quality information capable of improving the current management of weeds and fire in forests and woodlands in eastern Australia.

2. Impact of invasive African Olive (*Olea europea* ssp. *cuspidata*) on fuel load and structure in Cumberland Plain Woodland

2.1. Introduction

Plant invasions and fire can significantly change ecosystems (Fisher *et al.* 2009; Pyšek *et al.* 2012). Fuel can be defined as any burnable living and dead vegetation that may be consumed in the passage of a fire (Whelan 1995; Cochrane and Ryan 2009). Hence, the vegetation comprising a plant community provides the materials needed for fire. Fire influences and is influenced by plant community composition and structure resulting in complex relationships between fire behaviour and vegetation (Mandle *et al.* 2011). With the presence of weed species, these relationships may well change (Balch *et al.* 2013).

There are a number of characteristics of invasive plants that can influence fire behaviour or modify fire frequency in invaded areas. These include altering biomass accumulation through primary productivity differing from the native vegetation, shedding of leaves and branches that change the fuel load (Brooks *et al.* 2004), changes in rates of degradation and decomposition of shed material, changed flammability due to chemical composition that influences the ignition of plant material (Behm *et al.* 2004; Gill and Zylstra 2005) and structural changes in the vertical distribution of fuels (Pauchard *et al.* 2008; Berry *et al.* 2011).

The effects of woody weeds on fuel load and structure, fire regimes and intensity is poorly understood (Mandle *et al.* 2011). Several studies have quantified fire behaviour in invaded areas but most of these studies have been in grassland (Floyd 1966; D'Antonio and

Vitousek 1992; Briese 1996; Fine 2002; Brooks *et al.* 2004; Stohlgren and Schnase 2006; Pauchard *et al.* 2008; Burrows *et al.* 2009; Cohn *et al.* 2011). Although invasion by herbaceous weed species and the ways in which they change fire behaviour are beginning to be understood, the same is not true for woody weed species. Brooks *et al.* (2004) reviewed the mechanisms by which invasive plants alter fire regimes. Invasive plants can alter both intrinsic (moisture content of plant tissue and chemical composition of plant tissue) and extrinsic (fuel loads, fuel continuity and fuel packing ratio) fuel properties (Brooks *et al*. 2004). However, few studies have looked at how these plants alter fuel structure, load and flammability, particularly in an Australian context. Berry e*t al.* (2011) working in dry rainforests of Australia have shown that invasion by *Lantana camara* shifts the distribution of available fuels closer to the ground and provides a more continuous fuel layer in the understorey. Similarly, in coastal areas of northern Australia invaded by the shrub *Mimosa pigra*, the vegetation was dramatically altered in terms of fuel structure (Braithwaite *et al.* 1989).

The number of plant species introduced to Australia since European settlement is estimated to be about 25 000 (Groves 2002), with around 2700 non-native species registered as being naturalised (Groves *et al.* 2003). About 30% of these species represent a major threat to native plants including endangered and endemic flora (Groves *et al.* 2003). An unknown number are woody species grow in landscapes where fire is a regular disturbance and, at present, the effect of these species on fire behaviour remains largely unknown.

2.1.2 The invasive species African Olive (Olea europea ssp. cuspidata) and Cumberland Plain Woodland

African Olive (*Olea europea* ssp. *cuspidata*) was introduced from southern Africa to eastern Australia in the mid-19th century (Besnard *et al.* 2007), mainly as a hedge plant and rootstock for cultivated olives (NSW Scientific Committee 2010). It is now a well-established invasive plant in western and north western (e.g. Hunter Valley) regions of New South Wales (NSW) and in many other areas in South Australia and the Australian Capital Territory (Government of South Australia 2001; Cuneo and Leishman 2006; NSW Scientific Committee 2010). The seeds are spread by birds (Paton *et al.* 1988; Mladovan 1998) and have high viability, at least when the seed is young (Cuneo and Leishman 2012). In NSW, the African Olive is a declared noxious weed in 11 local government areas and has been declared as a key threatening process under the *NSW Threatened Species Conservation Act* 1995 (Cuneo and Leishman 2012; NSW Scientific Committee 2014). The control and management of African Olive is now needed in many conservation areas in NSW (Cuneo *et al.* 2009; NSW Scientific Committee 2010).

In areas that have been completely invaded, African Olive is a small to medium tree with a dense crown that prevents light reaching the understorey and consequently restricts the recruitment and growth of native plants (Cuneo and Leishman 2006; Cuneo and Leishman 2012). The invasion process takes place at a much slower rate than for most other weeds (Government of South Australia 2001). Seeds of African Olive have a resistant endocarp and endogenous dormancy making them slow to germinate (Rinaldi 2000), but this feature provides them the ability to remain in the soil seed bank for long periods (von Richter *et al.* 2005). Plants may require 5–10 years before they begin to produce fruit, however, individuals can live for many years and retain the ability to regenerate from stumps after felling or burning (Government of South Australia 2001). African Olive can form a climax vegetation that will dominate neighbouring vegetation unless management action takes place (Government of South Australia 2001; Cuneo and Leishman 2012).

Cumberland Plain Woodland originally covered an area of approximately 125 000 ha in the Sydney area, but due to extensive clearing has been reduced to about 10% of its original extent (Benson and Howell 2002). The small remaining fragments are threatened by changes in the fire regimes and invasion by exotic plants (Benson *et al.* 1990; Benson and Howell 2002; Cuneo and Leishman 2012). Consequently, it has been declared a Critically Endangered Ecological Community under the *NSW Threatened Species Conservation Act* 1995 and the *Commonwealth Environmental Protection and Biodiversity Conservation Act* 1999 (NSW National Parks and Wildlife Service 2002; Hill *et al.* 2005). It is composed mainly of open eucalypt woodland with an understorey dominated by grasses, forbs and scattered shrubs (Hill 2004; Hill *et al.* 2005). Understorey plants require high light incidence and specific environmental conditions for germination and subsequent growth (Benson and Howell 2002). The ecological effects of invasion of African Olive, including the disruption of light caused by the dense canopy cover, have recently been demonstrated by Cuneo and Leishman (2012). Although there have been several other studies of the impact of African Olive in Cumberland Plain Woodland (Cuneo and Leishman 2006; Cuneo *et al.* 2009; Cuneo and Leishman 2012), they have had an ecological focus and none of them have investigated the impact of this species on the fuel structure and flammability of this vegetation type.

The aim of this study was to determine the effect of invasion by African Olive on the load and structure of fuels in three types of vegetation: pristine Cumberland Plain Woodland (CPW), grassland in the initial stages of invasion (II) between 0 and 7 years of invasion, and long-term invasion (LI) with mature stands of African Olive from which all native vegetation has been displaced (15+ years of invasion). The flammability of fuels in the three areas was also investigated. To assess the differences among vegetation types the following research questions were formulated:

1. What are the differences amongst the fine fuel load and structure of mature stands of African Olive, Cumberland Plain Woodland and areas of initial invasion?

2. Is the flammability of African Olive different from vegetation in pristine Cumberland Plain Woodland and areas of initial invasion?

3. Does invasion by African Olive present a higher fire hazard to human communities and assets close to invaded sites?

In addition, to gain an understanding of the residence time of dead fine fuel, decomposition trials in each of the vegetation types were included. A combination of physical, chemical, and biological processes drive interactions among leaf litter from different species during decomposition (Graça *et al.* 2005). Mixing leaves from species with differing resource quality and leaf structure changes the chemical environment and physically alters the total litter surface where decomposition is occurring (Gartner and Cardon 2004).

2.2. Materials and methods

2.2.1. Study site

The Australian Botanic Garden, Mount Annan (34° 4'11.46"S, 150°46'1.82"E) located 56 km south west of Sydney (hereafter referred to as 'Mt Annan') was selected because of the long-term presence of African Olive trees and its proximity to remnant patches of Cumberland Plain Woodland (Cuneo et al. 2009). The general area including Mount Annan is composed of undulating plains and hills up to 300 m above sea level, dominated by fine textured clay yellow podzolic soils (Office of Environment and Heritage, 2014). The annual rainfall ranges from 600 to 922 mm and the climate is temperate (Cuneo et al. 2009). During the study period (2011–2013), the annual mean minimum temperature was 10.4 °C and the annual mean maximum temperature was 23.6 °C. Mean monthly rainfall is highest between February and March (approximately 95 mm) and lowest in September (approximately 43 mm)

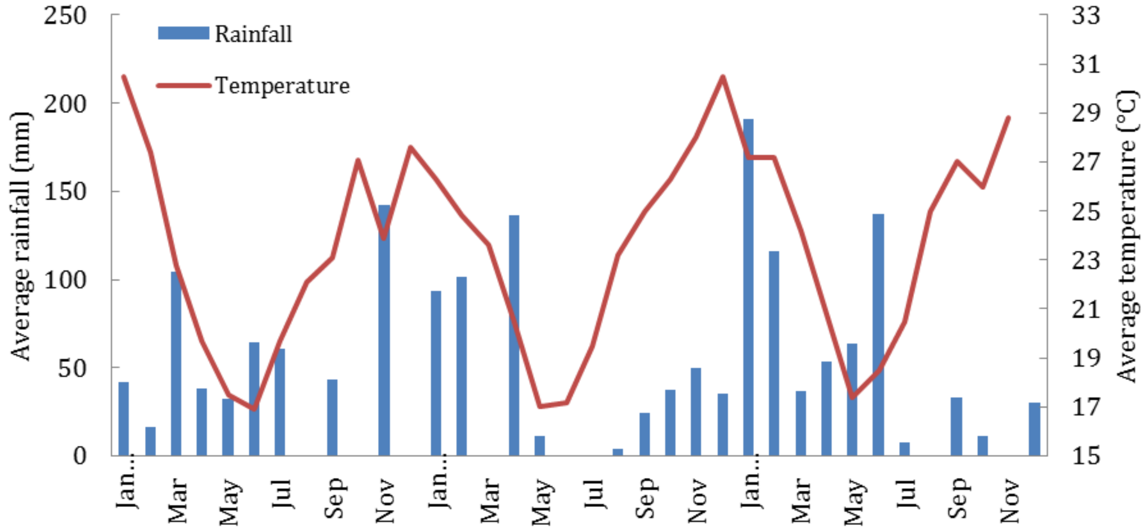

Figure 2. 1. Mean rainfall and temperature for 2011, 2012 and 2013 for Campbelltown weather station, NSW, 34.07°S, 150.82°E (Data obtained from Bureau of Meteorology - http://www.bom.gov.au/climate/data/stations/).

Mt Annan contains a small area of CPW which is managed due to its high conservation value (von Richter *et al.* 2005). Cumberland Plain Woodland is characterised by an open canopy composed of various species of *Eucalyptus* (e.g. *Eucalyptus moluccana*, *E. crebra*, *E. tereticornis*) and a grassy understorey with occasional shrubs, forbs and native grasses (e.g. *Bursaria spinosa*, *Themeda australis*, *Chloris ventricosa*, *Poa labillardieri*) (Benson 1992; Benson and Howell 2002; von Richter *et al.* 2005). Remnants of CPW at Mount Annan total about 10 ha and although it is considered to be well preserved, it has a history of partial clearing, grazing by domestic stock, and some localised cultivation and pasture improvement (Office of Environment and Heritage 2014).

2.2.2. Sampling design

Within Mt Annan, three areas were selected representing two invasion stages: initial invasion (II; 0-7 years of invasion) and long-term invasion (LI; 15+ years) as defined by Cuneo and Leishman (2006), and an area of remnant CPW (Figure 2.2). In each vegetation type, 50 × 50 m plots (n = 3) were established to investigate the fuel complex. Areas of long-term invasion (LI) have been infested with African Olive for about 15–20 years and represented an advanced stage of environmental degradation (see Cuneo and Leishman 2006). In areas of initial invasion (II), it is still possible to identify elements of the original vegetation (e.g. native grasses and shrubs, sapling eucalypts) amongst young African Olive trees and seedlings.

Three parallel transects of 50 m were established in each plot; two transects located 5 m away from the edges of the plot and one transect through the middle (i.e. at 25 m). For each transect there were 11 observation points spaced at 5 m intervals where the pin point intersect method was used (see below). To measure fuel hazard score, percentage cover score and fuel depth, five circles of 5 m radius were used at distances of 5, 15, 25, 35 and 45 m along each transect. Two quadrats of 1 × 1 m were randomly located along each transect to assess fine fuel biomass.

Figure 2. 2. Details of the three vegetation types used: (a) initial invasion (II) , (b) long-term invasion by African Olive (LI) and Cumberland Plain Woodland (CPW) in the Australian Botanic Garden, Mount Annan. Yellow arrows in (a) indicate young individuals of African Olive.

2.2.3. Measurement of fuel

Fuel architecture: pin point intersect method

The pin point intersect method (Canfield 1941) was used to determine fuel layer and stratum cover and height, gap fraction and gap size distribution. This method is commonly used to characterise horizontal fuel continuity, specifically gap size relative to fuel volume, as it is a feature that determines the likelihood that a flame front will self-sustain (Burrows *et al.* 2009). At each observation point a 2 m height pole was positioned vertically. The pole was marked to subdivide it into layers: 0–20, 20–50, 50–100, 100–150 and 150–200 cm above ground. For each observation point and layer, the number of times that 'live' and 'dead' fine fuel (any vegetation under 6 mm thickness) touched the pole was recorded. The live fine fuel was further classified as live grass, twigs, leaves and herbs. Dead fine fuel was classified as a single fuel type.

Fine fuel load and fuel moisture: destructive sampling method

Destructive sampling of the fuel was done using a 1 m^2 quadrat following an adaptation of the methodology used by Gould *et al.* (2007a). Two randomly located quadrats were established along each transect and the fine fuel (<6 mm thickness) in each layer was collected (litter, 0–20, 20-50, 50–100, 100–150, 150–200 cm heights), sorted according to state (live and dead) and weighed in the field with a digital scale to 0.01 g.

To measure fuel moisture content, live and dead samples were bulked separately in two height ranges, from 0–50 cm (low) and from 50–200 cm (high). After bulking the samples, subsamples were weighed using a field scale to 0.01 g. These subsamples were stored in paper bags and oven-dried for 48 h at 100 °C and re-weighed. The fuel moisture (%) of each sample was calculated as:

$$100\ x\ \frac{Wm - Dm}{Dm} \qquad [2.1]$$

where Wm is the fresh mass (g) and Dm is the dry mass (g). An average value for fuel moisture was calculated for height and state (live and dead) for each vegetation type and used to calculate the fine fuel biomass.

Visual scoring system

At regularly spaced points along each transect (i.e. 5, 15, 25, 35 and 45 m), a visual measurement of the hazard scoring system developed by Gould *et al*. (2007a) (hereafter described as the 'Vesta scoring system') was used to characterise fuel layers by estimating cover and hazard scores. The Vesta scoring system relies on visual estimation of aspects of vegetation such as cover, height and the proportion of dead material in different fuel layers.

Gould *et al.* (Gould *et al.* 2007a; Gould *et al.* 2011) identified five fuel layers that can be associated with fire behaviour. These layers can be broadly identified by change in bulk density and the following definitions were used as a guide in this study:

1. *"Overstorey tree and canopy layer – dominant and co-dominant trees forming the uppermost canopy layer of the forest. Trees are pole size (diameter at breast height over bark-dbhob 15–45 cm) or greater. The flammability of this layer depends primarily on the bark characteristics of the overstorey tree species, and the height*

and density of the forest. The bark type of different species can have a large impact on the rate of surface fuel accretion, transfer of a surface fire into the canopy and on the generation of firebrands. In this study the results for Fuel Hazard Score of the *overstorey tree and canopy layer* are represented as Bark and the Percentage Cover Score is represented as Canopy.

2. *Intermediate tree and canopy layer – shorter trees with crowns either below or extending into lower part of the forest canopy. These may be immature individuals of overstorey species or species of intermediate stature that form a distinct layer beneath the co-dominants of the overstorey (e.g.* Allocasuarina *spp.,* Banksia *spp.). Patches of eucalypt regrowth in the open or around scattered dominants may be classed as intermediate until they reach pole size. The intermediate tree layer can add a significant amount of bark fuel, and act as ladder fuel that carries fire into the overstorey canopy.*
3. *Elevated fuel layer – tall shrubs and other understorey plants without significant suspended material. This layer may include regeneration of the overstorey species intermixed with shrubs. Individual fuel components generally have an upright orientation, and include live and dead material.*
4. *Near-surface fuel layer – grasses, low shrubs, creepers, and collapsed understorey usually containing suspended leaf, twig and bark material from the overstorey vegetation. The height of this layer can vary from just centimetres to over a metre above the ground. Fuel layer components typically have a mixed orientation ranging from horizontal to vertical, and the layer is capable of suspending leaves, twigs and bark above the ground.*

5. *Surface fuel layer – leaves, twigs and bark of overstorey and understorey plants. Fuel components are generally horizontally layered. This layer usually makes up the bulk of the fuel consumed and provides most of the energy released by a fire. Surface fuel burns by both flaming and smouldering combustion, and determines the flame depth of a surface fire. The duff layer of decomposed litter fuel is absent in most dry eucalypt forests."* (Gould *et al*. 2011, p. 17–19).

The overall fuel hazard was rated using a categorical score from 0 to 4 based on visual assessment of the percent cover score (PCS) and the fuel hazard score (FHS) for each of the five fuel layers according to Gould *et al*. (2011). At each point used for visual scoring of fuel load, litter depth and heights of near surface and elevated fuel layers were measured using a ruler or tape measure according to the method of McCarthy *et al.* (1999).

2.2.4. Soil sampling

Five soil samples were collected at two depths (0–5 and 5–10 cm) along each transect using a steel core (5 cm diameter × 10 cm depth) with a bevelled edge and a small diameter auger. Soils were sieved to 4 mm and 2 mm in the field to remove rocks and debris. For each depth, the samples were bulked and a composite sample representing each transect was formed. This generated three samples per plot and nine soil samples per vegetation type. The samples were stored in sealable plastic bags at <5 °C until further analysis. An additional soil core (0–10 cm) was taken in each plot and kept intact for determination of soil bulk density.

Soil pH was measured in water suspension (1:2; soil:H_2O) using a pH meter (pH Cube, TPS, Australia). Fresh soil (approximately 7.5 g) was mixed with 15 ml of deionised water and shaken on a wheel rotator for 15 min. Samples were allowed to settle for 15 min before measurement. Intact soil cores were weighed while fresh then dried to constant weight at 105 °C and values were used to calculate bulk density and gravimetric water content (Loveday 1974). The bulk density (BD) and gravimetric water content (WCg) are calculated as:

$$\text{BD (g cm}^{-3}\text{)} = \text{Mass of dry soil (g)/ Volume of core (cm}^{3}\text{)} \qquad [2.2]$$

$$\text{WCg (g g}^{-1}\text{)} = \text{Mass of wet soil} - \text{Mass of dry soil (g)/ Mass of dry soil (g)} \qquad [2.3]$$

Soil samples were oven dried at 105 °C until constant weight and ground to a fine powder in a mortar grinder (MZ1000, RETSCH, Germany) and analysed for total carbon (%C) and nitrogen (%N) by dry combustion (Elementar Vario Max CNS Analysensysteme GmbH, Hanau, Germany).

2.2.5 Decomposition experiment

Leaf litter from two vegetation types (CPW and LI) was collected from the surface fuel layer. This material was sieved to 2 mm to remove soil and debris associated with its collection, air-dried at room temperature and stored in a humidity-controlled environment until required. The common agricultural legume, Lucerne (*Medicago sativa*) was chosen as an alternative litter type and was obtained from a local commercial supplier. Litter bags were made by sewing together two pieces (15 × 15 cm squares) of 70% nylon shadecloth on three sides with cotton thread. Additional bags (Control) with no contents were sewn together on four sides. Portions of leaf litter and Lucerne were weighed (approximately 5 g) and distributed equally inside the litter bags to form a continuous layer and an aluminium label was added for identification. Litter bags were closed on the open side with aluminium staples.

Litter bags containing each of four litter treatments (African Olive leaves (referred to as 'African Olive'), eucalypt leaves (referred to as 'Eucalyptus'), 50% of each of these two litter types (referred to as 'Mix') and stalks and leaves of Lucerne (referred to as 'Lucerne') and empty bags (referred to as 'Control') were placed in the three vegetation types (CPW, II and LI). At each location (n = 3), there were three time point replicates (n = 3) of each of the three treatment/Control (n = 5). The bags were sampled at the beginning of the decomposition experiment (t = 0), and 6 months (t = 6), 12 months (t = 12) and 17 months (t = 17). For each time point, there were five replicates of each treatment/Control (total number of litter bags = 225). To minimise soil disturbance and to simulate decomposition conditions in the surface fuel layer, litter bags were placed on the surface of the soil or litter and anchored at one corner using an aluminum tent peg (Fig. 2.3). Litter bags were grouped together with one replicate from each treatment and a Control.

At each sampling time, the litter bags were collected, taken to the laboratory and any adhering dirt or litter was carefully brushed off. The contents of each litter bag were removed and re-bagged in paper bags to prevent loss of litter fragments, air-dried at room temperature and weighed. The samples were ground to a fine powder in a mortar grinder (MZ1000, RETSCH, Germany) and analysed for total carbon (%C) and nitrogen (%N) by dry combustion (Elementar Vario Max CNS Analysensysteme GmbH, Hanau, Germany).

Figure 2. 3. An example of the arrangement and positioning of decomposition bags at the Australian Botanical Garden, Mount Annan.

The decomposition constant (k) was calculated from a first-order exponential decay equation (Olson 1963; Harmon *et al.* 1999):

$$M_t = M_0 \times e^{(-kt)} \quad [2.4]$$

where M_t is litter mass at time t and M_0 is litter mass at time 0. Rearranged, Equation 2.4 can also be expressed as a log-linear equation:

$$\mathrm{Ln}\left(\frac{M_t}{M_0}\right) = -kt \qquad [2.5]$$

Or to obtain the value k:

$$k = -\left[\frac{\ln\left(\frac{M_t}{M_0}\right)}{\mathrm{t}}\right] \qquad [2.6]$$

2.2.6. Assessment of fuel flammability

Flammability was assessed using a mass-loss calorimeter (MLC; Fire Testing Technology; UK). The MLC consists of a conical heater capable of producing radiative fluxes between 10 and 100 kW m^{-2} and a load cell to measure the change in mass of a sample over time. The cone heater and load cell are contained within a stainless steel enclosure, which is supplied with compressed air at a flow rate of 140 L min^{-1}. A 60 cm stainless steel chimney on top of the enclosure contains thermocouples that are calibrated using high purity (99%) methane gas (BOC Ltd, North Ryde, NSW, Australia) to quantify heat release as described in the standard ISO 13927 (ISO, 2001). In the MLC, a sample holder (10 ×10 ×

5 cm) with a porosity of 27% was used to allow diffusion of air through the fuel samples (Possell and Bell 2013).

After measuring the fuel moisture associated with destructive sampling (see Section 2.2.3), a sub-sample from each fuel layer per transect was combusted. This generated nine samples from each layer (litter, dead low, dead high, live low and live high) per vegetation type and a total of 45 samples per vegetation type. The weight of each sample varied according to the fuel type. The fuel samples were trimmed to fit the holder to uniformly cover the exposed surface area and sample thickness was maintained at 5 cm. Burns were done using an irradiance of 25 kW m^{-2} and a 10 kV spark igniter was used to provide piloted ignition. Cruz *et al*. (2011) and Silvani *et al*. (2009) indicate that irradiances of 25 kW m^{-2} are achievable during a natural fire at the fire front and can remain high for some time once the front has passed. This period of time is comparable to the length of time each burn was conducted (200 to 600 s). Heat rate release (HRR; kW m^{-2}) and mass loss rate (MLR; g s^{-1}) were recorded at 1 Hz and the time-to-ignition (TTI; s) and flame duration (FD; s) was recorded manually. The average effective heat of flaming combustion (AEHC; MJ kg^{-1}) was calculated as the total heat release divided by the mass loss (MLCCalc; Fire Testing Technology, UK).

Outputs from the MLC were related to the components of flammability (see Chapter 1) as defined by Anderson (1970) and Martin *et al*. (1994). Ignitability was determined by measuring the time-to-ignition; sustainability was assessed from the duration of flaming combustion; combustibility was considered to be equivalent to the mass-loss rate (burning rate) and; consumability was regarded as the residual mass fraction of the material burnt.

2.2.7. Statistical analyses

The amount and structure of fuel in the three vegetation types (vegetation architecture; CPW, II and LI) was compared using one-way ANOVA with Tukey's Honest Significant Difference (HSD) post-hoc tests. The point intersect data was square root transformed and (i.e. the number of touches in each layer and the total number of touches amongst vegetation types) were compared for each fuel class (live fine fuel: grass, twigs/leaves and herbs, and dead fine fuel) amongst vegetation types for each individual layers separately. Values for fine fuel biomass collected during destructive sampling were log-transformed to normalise the data prior to one-way ANOVA. Fuel depth and soil properties were compared amongst vegetation types within each layer using one-way ANOVA with Tukey's HSD tests. Values derived from the visual scoring system (i.e. overall FHS, and FHS and PCS for the five fuel layers (overstorey tree and canopy; intermediate tree and canopy; elevated; near surface and surface) were compared using one-way ANOVA with Tukey's HSD tests.

The proportion mass loss (%ML), change in the total carbon (%C), change in total nitrogen (%N), change in C:N and decomposition constant (k) were modelled separately. The overall effect of time for each treatment was tested using Repeated measures (RM) ANOVA with a Mauchly's test of Sphericity. Greenhouse-Geisser and Huynh-Feldt correction factors were applied when the sphericity assumption was not achieved. A Tukey's HSD post-hoc test was used in all cases. Differences among treatments within one time point were tested using one-way ANOVA combined with Tukey's HSD tests.

Data for each flammability component (HRR, MLR, TTI, FD) were compared by using one-way ANOVA with Tukey's HSD tests. In order to meet the criteria for using one-way

ANOVA test the values of TTI and FD were log-transformed and the MLR was arc sin-transformed. All statistical analyses were conducted in the software SPSS version 22.

2.3. Results

2.3.1. Fuel architecture

The arrangement of the fuel differed considerably among the three vegetation types, at least in the lower fuel layers. Areas with long-term invasion by African Olive (LI) had a significantly smaller number of total touches than Cumberland Plain Woodland (CPW; one-way ANOVA, Tukey's HSD test; $P = 0.015$; Table 2.1) and areas representing initial invasion by African Olive (II; $P < 0.001$). In the layer represented by 0–20 cm, II had a significantly greater total number of touches than CPW (one-way ANOVA, Tukey's HSD test; $P = 0.003$) and LI ($P < 0.001$). The layer represented by 20–50 cm, II had a significantly greater total amount of touches than CPW and LI (one-way ANOVA, Tukey's HSD test; $P < 0.001$), and LI had significantly fewer touches than CPW ($P = 0.032$). For the fuel layer represented by 50–100 cm, the total number of touches for II was greater than for CPW (one-way ANOVA, Tukey's HSD test; $P < 0.001$) and LI ($P = 0.016$), and there was no significant difference between CPW and LI ($P = 0.497$). At 100–150 cm, the total number of touches for LI was not statistically different when compared to II (one-way ANOVA, Tukey's HSD test; $P = 0.977$) but both were statistically larger than CPW ($P = 0.006$ and $P = 0.003$, respectively). For the fuel layer represented by 150–200 cm, the total number of touches for LI was greater than for II (one-way ANOVA, Tukey's HSD; $P = 0.003$) and CPW ($P < 0.001$). The total number of touches for CPW was statistically fewer than II ($P < 0.001$; Table 2.1).

Table 2.1. Total number of touches (± standard deviation) in each fuel height layer (0-20, 20-50, 50-100, 100-150, 150-200 cm and total) for three vegetation types (Cumberland Plain Woodland (CPW), initial invasion (II) and long-term African Olive invasion (LI)) at the Australian Botanical Garden, Mount Annan. Statistical comparisons (one-way ANOVA and post-hoc Tukey's HSD tests) were made among vegetation types within each layer. Different letters represent significant statistical differences.

	Vegetation type		
Height (cm)	CPW	II	LI
150–200	2 ± 1$^{\mathrm{a}}$	34 ± 12$^{\mathrm{b}}$	40 ± 12$^{\mathrm{c}}$
100–150	7 ± 3^{a}	27 ± 10^{b}	20 ± 7a^{b}
50–100	14 ± 3$^{\mathrm{A}}$	45 ± 9$^{\mathrm{B}}$	18 ± 5$^{\mathrm{A}}$
20–50	30 ± 7^{A}	127 ± 29^{B}	7 ± 2^{C}
0–20	145 ± 26$^{\underline{\mathrm{a}}}$	256 ± 54$^{\underline{\mathrm{b}}}$	13 ± 3$^{\underline{\mathrm{c}}}$
Total	198 ± 17$^{\underline{a}}$	488 ± 34$^{\underline{b}}$	97 ± 7$^{\underline{c}}$

Most touches of fuel (73%) in Cumberland Plain Woodland areas occurred in the layer closest to the ground (0–20 cm; Table 2.2). In this layer, approximately 35% of the fuel was composed of live grass (LG), 40% was composed of dead fine fuel (DFF) and approximately 25% was composed of live herbs (H). There was no statistical differences between the number of touches of live grass and dead fine fuel (one-way ANOVA, Tukey's HSD test; $P = 0.984$), however, live grass and dead fine fuel had a greater number of touches than live herbs ($P < 0.001$ for both).

Areas of initial invasion (II) had more than half of the touches (52%) of the fuel in the lowest layer from 0–20 cm, a quarter of touches (25%) in the 20–50 cm layer and an even distribution shared between the other three height classes (Table 2.2). The lowest layer from 0–20 cm was dominated by dead fine fuel, with about half (51%) of the total number of touches. Live grass was the second most abundant fuel type representing around 30% of the fuel composition in this layer. Dead fine fuel had a significantly greater number of touches than live grass and live herbs at the bottom layer (one-way ANOVA, Tukey's HSD test; $P = 0.027$ and $P < 0.001$, respectively). The uppermost layers, 100–150 cm and 150–200 cm, were dominated by live twigs mostly due to branches from young individuals of African Olive (Table 2.2).

Areas of long-term invasion by African Olive (LI) had a very different structure in terms of total number of touches for each layer when compared to the other two vegetation types. Areas invaded by African Olive had the greatest proportion of fuel distributed with the top three layers (i.e. from 50 to 200 cm) representing approximately 80% of the total number of touches. The two uppermost layers, 100–150 cm and 150–200 cm, were composed only of live twigs and dead fine fuel (mainly composed of dead twigs less than 5 mm diameter thick). From 100–150 cm, dead fine fuel contributed the greatest proportion of the fuel (73% of touches) and was statistically different from live twigs (one-way ANOVA, Tukey's HSD test; $P < 0.001$). The uppermost layer, from 150–200 cm, was characterised by dead fine fuel (60% of touches) and live twigs (40%) with no statistical difference between these two components (one-way ANOVA, Tukey's HSD test; $P = 0.264$).

Table 2.2. Total number of touches (mean ± standard deviation) of each class (live grass (LG), live herbs (H), live leaves and twigs (LTL) and dead fine fuel (DFF) in each fuel layer (0–20, 20–50, 50–100, 100–150, 150–200 cm) for three vegetation types (Cumberland Plain Woodland (CPW), initial invasion (II) and long-term African Olive invasion (LI) at the Australian Botanical Garden, Mount Annan. Statistical comparisons (one-way ANOVA and post-hoc Tukey's HSD tests) were made among fuel types within the same height class and vegetation type. Different letters represent significant statistical differences.

	Fuel and vegetation type											
	CPW				II				LI			
Fuel layer (cm)	LG	H	LTL	DFF	LG	H	LTL	DFF	LG	H	LTL	DFF
150–200	0^{a}	0^{a}	1 ± 2^{a}	1 ± 2^{a}	0^{A}	0^{A}	26 ± 12^{B}	7 ± 2^{A}	0^{a}	0^{a}	16 ± 2^{b}	24 ± 4^{a}
100–150	0^{a}	0^{a}	5 ± 6^{b}	2 ± 2^{a}	1 ± 1^{A}	0^{A}	21 ± 10^{B}	6 ± 2^{A}	0^{a}	0^{a}	5 ± 2^{b}	14 ± 4^{a}
50–100	2 ± 3^{ab}	0^{a}	7 ± 3^{b}	5 ± 2^{b}	7 ± 10^{AB}	1 ± 1^{A}	22 ± 12^{B}	15 ± 11^{B}	0^{ab}	0^{a}	7 ± 4^{b}	11 ± 3^{b}
20–50	9 ± 7b^{ab}	2 ± 1^{c}	2 ± 1^{ac}	17 ± 16^{b}	40 ± 43^{A}	9 ± 2^{B}	9 ± 9^{B}	69 ± 79^{A}	0^{a}	0^{b}	3 ± 4^{b}	4 ± 2^{a}
0–20	52 ± 21^{a}	35 ± 4^{b}	0^{c}	58 ± 26^{a}	75 ± 54^{A}	48 ± 10^{B}	1 ± 1^{B}	131 ± 135^{C}	0^{a}	3 ± 1^{b}	6 ± 2^{b}	3 ± 3^{c}

2.3.2. Fine fuel load

There were no significant differences among total fine fuel biomass of the three vegetation types (Table 2.3). Areas of long-term invasion by African Olive (LI) had significantly greater dead fine fuel biomass than CPW or II (one-way ANOVA, Tukey's HSD test; P = 0.010), however there was no significant difference in total dead fine fuel biomass between CPW and II (P = 0.555). When fine fuel biomass was compared according to layer and class (i.e. dead or live fuel), LI had a significantly greater amount of litter (one-way ANOVA, Tukey's HSD test; P = 0.002) compared to both II and CPW. There were no significant differences in litter biomass between CPW and II (P = 0.049). Long-term invaded areas had significantly smaller total live biomass (one-way ANOVA, Tukey's HSD test; P = 0.001) than CPW and there was no significant difference in total live biomass of LI compared to II (P = 1.000). The total live biomass in CPW was significantly smaller than II (one-way ANOVA, Tukey's HSD test; P <0.001).

Table 2.3. Fine fuel biomass ((kg m^{-2}); mean ± standard deviation) in classes (live, dead and total) and fuel height layer (litter, 0–20, 20–50, 50–100, 100–150, 150–200 cm and total) for three vegetation types (Cumberland Plain Woodland (CPW), initial invasion (II) and long-term African Olive invasion (LI)) at the Australian Botanical Garden, Mount Annan. Statistical comparisons (one-way ANOVA and post-hoc Tukey's HSD tests) were made among vegetation types within each layer. Different letters represent significant statistical differences among the logarithmic transformed data.

	Fuel and vegetation type								
	Live fine fuel			Dead fine fuel			Total		
Fuel layer (cm)	CPW	II	LI	CPW	II	LI	CPW	II	LI
150–200	0.00 ± 0.00^{a}	0.03 ± 0.03^{a}	0.00 ± 0.00^{a}	0.00 ± 0.01^{a}	0.00 ± 0.01^{a}	0.04 ± 0.04^{a}	0.00 ± 0.00^{A}	0.02 ± 0.03^{A}	0.02 ± 0.03^{A}
100–150	0.00 ± 0.00^{a}	0.03 ± 0.03^{a}	0.00 ± 0.00^{a}	0.00 ± 0.01^{a}	0.01 ± 0.01^{a}	0.02 ± 0.03^{a}	0.00 ± 0.00^{A}	0.02 ± 0.02^{A}	0.01 ± 0.02^{A}
50–100	0.00 ± 0.00^{a}	0.03 ± 0.03^{a}	0.00 ± 0.00^{a}	0.00 ± 0.01^{a}	0.01 ± 0.01^{a}	0.01 ± 0.01^{a}	0.00 ± 0.01^{A}	0.02 ± 0.02^{A}	0.00 ± 0.01^{A}
20–50	0.00 ± 0.00^{a}	0.02 ± 0.02^{a}	0.00 ± 0.00^{a}	0.00 ± 0.01^{a}	0.00 ± 0.00^{a}	0.00 ± 0.00^{a}	0.00 ± 0.00^{A}	0.01 ± 0.01^{A}	0.00 ± 0.00^{A}
0–20	0.02 ± 0.02^{a}	0.04 ± 0.04^{a}	0.00 ± 0.01^{a}	0.05 ± 0.03^{a}	0.03 ± 0.03^{a}	0.00 ± 0.01^{a}	0.04 ± 0.03^{A}	0.04 ± 0.03^{A}	0.00 ± 0.00^{A}
Litter	-	-	-	0.43 ± 0.24^{a}	0.33 ± 0.36^{a}	0.73 ± 0.52^{b}	0.43 ± 0.25^{A}	0.33 ± 0.36^{A}	0.72 ± 0.52^{B}
Total	0.05 ± 0.01^{a}	0.17 ± 0.03^{b}	0.02 ± 0.00^{b}	0.51 ± 0.12^{a}	0.41 ± 0.09^{a}	0.83 ± 0.20^{b}	0.57 ± 0.07^{A}	0.58 ± 0.06^{A}	0.85± 0.12^{A}

2.3.3. Visual scoring of fuel and fuel depth

The three vegetation types had significantly different (one-way ANOVA, Tukey's HSD test; P <0.001) fuel hazard scores (FHS) for the surface fuel layer (Table 2.4) with LI having the highest score, followed by CPW and II. For the near-surface fuel layer, LI had a significantly lower FHS (one-way ANOVA, Tukey's HSD test; P <0.001) when compared with CPW and II. There was no statistical difference (one-way ANOVA, Tukey's HSD test; P = 0.195) between CPW and II for the near-surface fuel layer. Long-term invaded sites had a significantly higher FHS (one-way ANOVA, Tukey's HSD test; P <0.001) for the elevated fuel layer compared to CPW and II but there was no statistical difference between CPW and II for this fuel layer. The bark fuel of II was significantly lower (one-way ANOVA, Tukey's HSD test; P <0.001) than LI and CPW and there was no statistical difference between CPW and II (P = 0.692).

Table 2.4. Fuel hazard score (mean ± standard deviation) in each fuel layer (Surface, Near-surface, Elevated and Bark) for three different vegetation types (Cumberland Plain Woodland (CPW), initial invasion (II) and long-term African Olive invasion (LI)) at the Australian Botanical Garden, Mount Annan. Statistical comparisons were made among vegetation types within the same fuel layer. Different letters represent significant statistical differences.

	Vegetation type		
Fuel layer	CPW	II	LI
Surface	2.0 ± 0.2[a]	1.2 ± 0.2[b]	3.2 ± 0.2[c]
Near-surface	2.5 ± 0.3[A]	2.8 ± 1.4[A]	0.9 ± 0.2[B]
Elevated	1.9 ± 0.2[*a*]	1.8 ± 0.5[*a*]	2.8 ± 0.6[*b*]
Bark	2.4 ± 0.2[*A*]	0.4 ± 0.3[*B*]	2.3 ± 0.1[*A*]

The percentage cover score (PCS) also varied among fuel layers and between vegetation types. For the surface fuel layer, LI had a higher PCS than CPW (one-way ANOVA, Tukey's HSD test; P = 0.018) and II (P <0.001), and CPW a greater PCS than II (P <0.001) in this layer. For the near surface layer, the PCS for LI was significantly smaller than CPW and II (one-way ANOVA, Tukey's HSD test; P <0.001), but there was no difference between CPW and II (P = 345). For the elevated fuel layer, LI had a significantly greater PCS than CPW and II (one-way ANOVA, Tukey's HSD test; P <0.001) and there was no statistical difference between CPW and II (P = 1.000). The PCS for the canopy layer was different amongst the three vegetation types (one-way ANOVA, Tukey's HSD test; P = 0.001).

Table 2.5. Percentage cover score (mean ± standard deviation) for each fuel layer (Surface, Near-surface, Elevated and Bark) for three vegetation types (Cumberland Plain Woodland (CPW), initial invasion (II) and long-term African Olive invasion (LI)) at the Australian Botanical Garden, Mount Annan. Statistical comparisons were made among vegetation types within the same fuel layer. Different letters represent significant statistical differences.

	Vegetation type		
Fuel layer	CPW	II	LI
Surface	2.4 ± 0.3^{a}	1.4 ± 0.5^{a}	3.0 ± 0.7^{b}
Near-surface	2.7 ± 0.4^{A}	3.0 ± 1.1^{A}	1.3 ± 0.4^{B}
Elevated	1.9 ± 0.1^{a}	1.9 ± 0.5^{a}	2.8 ± 0.5^{b}
Canopy	2.2 ± 0.4$^{\underline{a}}$	0.5 ± 0.3$^{\underline{b}}$	3.9 ± 0.2$^{\underline{c}}$

The depth of the surface fuel layer of LI was statistically greater than CPW and II (one-way ANOVA, Tukey's HSD test; P <0.001). The height of near-surface fuel layer in CPW and LI was smaller than in II (one-way ANOVA, Tukey's HSD test; P <0.001) mainly due to a prominent grassy layer in II. The height of the elevated fuel layer in LI was statistically greater than in CPW (one-way ANOVA, Tukey's HSD test; P = 0.036) and II (P = 0.011).

Table 2.6. Fuel depth (mean ± standard deviation) for each fuel layer (Surface, Near-surface, Elevated and Canopy) in three vegetation types (Cumberland Plain Woodland (CPW), initial invasion areas (II) and long-term African Olive invasion (LI)) at the Australian Botanical Garden, Mount Annan. Statistical comparisons were made among fuel types within the same fuel layer. Different letters represent significant statistical differences.

	Vegetation type		
Fuel layer	LI	II	CPW
Surface (mm)	22.9 ± 2.0[a]	6.9 ± 01.0[b]	13.6 ± 0.4[c]
Near-surface (cm)	19.1 ± 5.0[a]	36.4 ± 14.8[b]	24.4 ± 5.4[a]
Elevated (m)	2.8 ± 0.5[a]	2.0 ± 00.2[b]	1.9 ± 0.1[b]

2.3.4. Soil composition

The soil pH was similar among all vegetation types (one-way ANOVA, Tukey's HSD test; P >0.05; Table 2.7). Similarly, there was no statistical difference for total N and total C in the upper soil layer (0–5 cm depth) amongst the vegetation types. For the deeper layer (5–10 cm depth), total N of soil from CPW was statistically lower than both LI and II (one-way ANOVA, Tukey's HSD test; P <0.001 for both). Total C in soil from CPW was statistically lower than LI (one-way ANOVA, Tukey's HSD test; P = 0.020) and II (P = 0.002). Consequently, Cumberland Plain Woodlands had a significantly higher C:N ratio than LI (one-way ANOVA, Tukey's HSD test; P <0.014) and II (P <0.005).

Table 2.7. Bulk density, gravimetric water content, soil pH, total N, total C and C:N ratio (mean ± standard deviation) for three vegetation types (Cumberland Plain Woodland (CPW), initial invasion (II) and long-term African Olive invasion (LI)) at the Australian Botanical Garden, Mount Annan. Statistical comparisons were made among vegetation types. Different letters represent significant statistical differences.

		Vegetation type		
		CPW	II	LI
0 to 5 cm	Bulk density (g cm^{3})	0.11 ± 0.01	0.10 ± 0.04	0.10 ± 0.03
	Gravimetric water content (g g^{-1})	92.8 ± 17.2	91.4 ± 49.4	91.8 ± 30.3
	pH	5.8 ± $0.3^{\underline{a}}$	6.0 ± $0.2^{\underline{a}}$	6.5 ± $0.3^{\underline{a}}$
	Total N (%)	0.37 ± 0.11^{a}	0.38 ± 0.07^{a}	0.42 ± 0.09^{a}
	Total C (%)	5.42 ± 1.39^{A}	5.03 ± 0.95^{A}	5.61 ± 1.31^{A}
	C:N ratio	14.6 ± 1.02^{a}	13.1 ± 0.78^{b}	13.3 ± 0.92^{b}
5 to 10 cm	Bulk density (g cm^{3})	0.14 ± 0.01	0.15 ± 0.05	0.16 ± 0.04
	Gravimetric water content (g g^{-1})	120.0 ± 31.5	134.3 ± 12.8	136.8 ± 17.7
	pH	5.7 ± $0.3^{\underline{a}}$	6.0 ± $0.1^{\underline{a}}$	6.1 ± $0.2^{\underline{a}}$
	Total N (%)	0.18 ± 0.04^{a}	0.30 ± 0.03^{b}	0.28 ± 0.04^{b}
	Total C (%)	2.88 ± 0.67^{A}	4.07 ± 0.47^{B}	3.81 ± 0.75^{B}
	C:N ratio	15.4 ± 1.27^{a}	13.1 ± 0.78^{b}	13.3 ± 1.03^{b}

2.3.5. Decomposition experiment

Loss of mass

The mass decay of the different treatments followed the same pattern of exponential decay in all vegetation types (Figure 2.4). Overall, after 6, 12 and 17 months of incubation, all of the litter bags retrieved from the field had lost some matter after each time period, regardless of treatment or vegetation type (RM ANOVA, Tukey's HSD test; $F = 546.1$, $P < 0.001$). There was a significant overall effect of time and treatment for LI (RM ANOVA, Tukey's HSD test; $F = 20.286$, $P < 0.001$), II ($F = 20.758$, $P < 0.001$) and CPW ($F = 6.028$, $P < 0.001$).

Change in total carbon and nitrogen

The initial chemical composition of the four types of litter or plant material used is shown in Table 2.8. The C and N content of the four treatments at time 0 ranged from 40–50% and 1–3%, respectively, with significant difference among all treatments (one-way ANOVA, Tukey's HSD test; P <0.001). Treatments involving a mix of leaves (Mix) had the greatest proportion of total C with Lucerne having the lowest. Total N of the treatments showed the opposite pattern with Lucerne having the greatest proportion of N. This was also reflected in the C:N ratio.

Table 2.8. Initial (t = 0) chemical composition of litter or plant material used in the four treatments. Data represent the mean ± standard deviation for five replicate analyses from bulked litter or plant material. Different letters represent significant statistical differences.

Treatment	Lucerne	Eucalyptus	African Olive	Mix
%N	2.72 ± 0.02^{a}	1.61 ± 0.45^{b}	0.99 ± 0.39^{c}	1.30 ± 0.21^{d}
%C	41.55 ± 0.50^{A}	47.78 ± 2.11^{B}	47.09 ± 5.10^{C}	49.09 ± 5.92^{D}
C:N	15.25 ± 01.30^{a}	29.74 ± 11.14^{b}	47.68 ± 11.18^{c}	37.71 ± 0.57^{d}

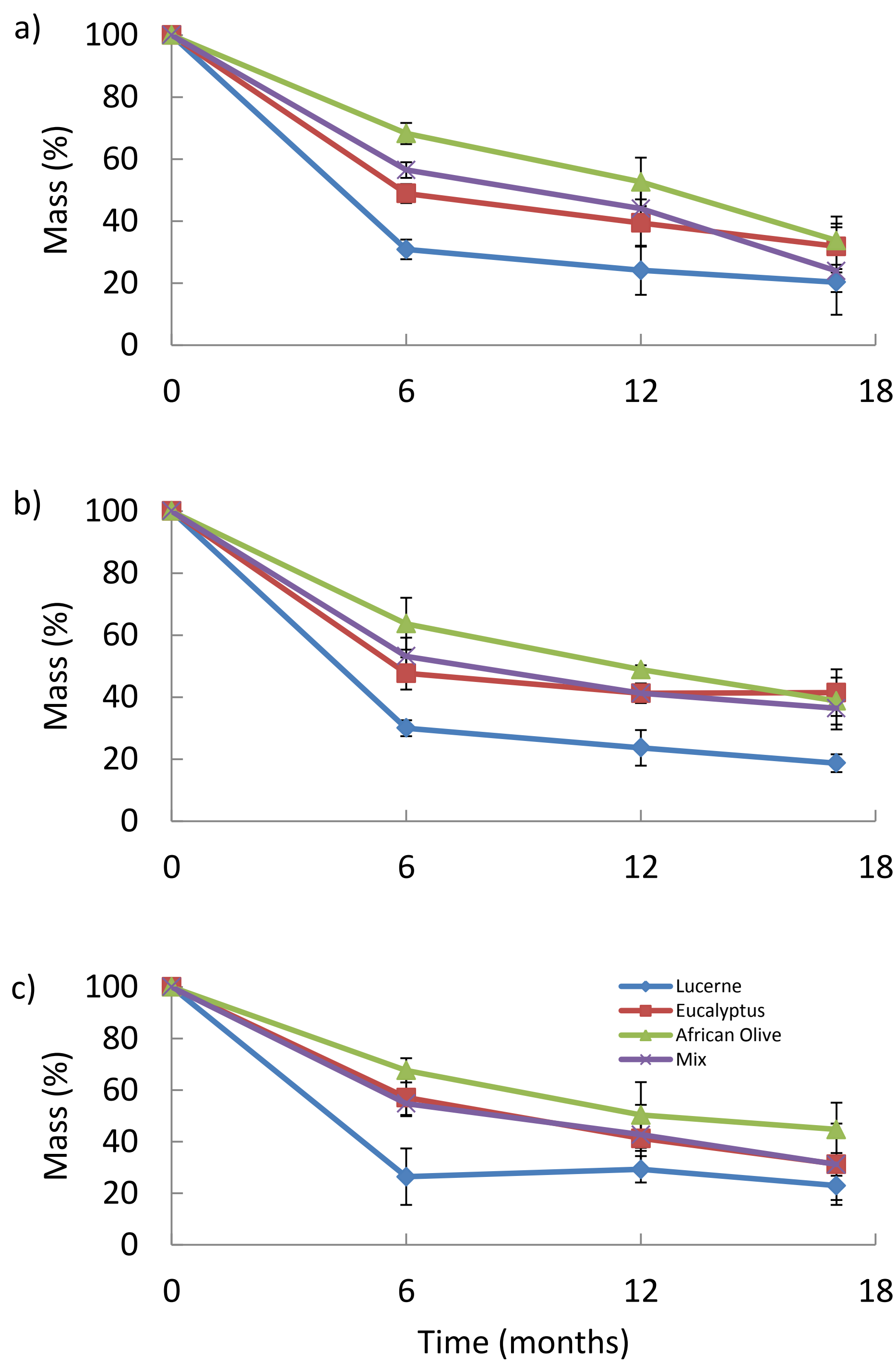

Figure 2. 4. Mass loss (%) of four treatments (Lucerne, Eucalyptus, African Olive and Mix) after 6, 12 and 17 months of incubation in (a) Cumberland Plain Woodland (CPW), (b) initial invasion (II) and (c) long-term African Olive invasion (LI) at the Australian Botanic Garden, Mount Annan. The control bags maintained their weight over time and are not included.

After 6 months of incubation, the C content of the treatments ranged between 40 and 50% for all the vegetation types (Table 2.9). Overall, there was no statistical difference in total C among treatments in LI and CPW (one-way ANOVA, Tukey's HSD test; $P > 0.05$). Treatments from II were statistically different with African Olive litter having a smaller proportion of total C than Eucalyptus (one-way ANOVA, Tukey's HSD test; $P < 0.001$) and Mix ($P = 0.011$) treatments. Eucalyptus litter had a greater proportion of C than Mix (one-way ANOVA, Tukey's HSD test; $P = 0.006$) and Lucerne ($P = 0.010$).

After 12 months of incubation, the C content of treatments in LI showed significant differences. The total C of Eucalyptus litter was significantly greater than Lucerne, African Olive and Mix (one-way ANOVA, Tukey's HSD test; $P < 0.001$ for all comparisons). Overall, the treatments from II and CPW showed no significant difference in total C (one-way ANOVA, Tukey', HSD test; $P > 0.05$) after 12 months of incubation indicating stabilisation in C loss. Similar patterns were found after 17 months incubation and the C content of all treatments in the three vegetation types was still between 40 and 50%.

Table 2.9. Carbon content (%, mean ± standard deviation) of four decomposition treatments (Lucerne, Eucalyptus, African Olive and Mix) after 6, 12 and 17 months of incubation for three vegetation types (Cumberland Plain Woodland (CPW), initial invasion (II) and long-term African Olive invasion (LI)) at the Australian Botanical Garden, Mount Annan. Different letters represent significant statistical differences. The comparisons were made within one vegetation type at each time between treatments.

		Incubation time (months)		
Vegetation type	Treatment	6	12	17
CPW	Lucerne	42.93 ± 3.84$^{\mathrm{a}}$	46.23 ± 1.79$^{\underline{\mathrm{a}}}$	41.87 ± 7.83^{ab}
	Eucalyptus	48.12 ± 3.31$^{\mathrm{a}}$	48.58 ± 2.07$^{\underline{\mathrm{a}}}$	46.05 ± 5.83^{c}
	African Olive	42.16 ± 3.35$^{\mathrm{a}}$	42.88 ± 1.36$^{\underline{\mathrm{a}}}$	43.47 ± 1.43^{a}
	Mix	45.11 ± 3.29$^{\mathrm{a}}$	44.62 ± 2.50$^{\underline{\mathrm{a}}}$	43.97 ± 2.49^{b}
II	Lucerne	42.78 ± 3.75$^{\mathrm{AB}}$	46.27 ± 1.72$^{\mathrm{a}}$	41.71 ± 7.75$^{\underline{\mathrm{a}}}$
	Eucalyptus	46.07 ± 4.18$^{\mathrm{C}}$	48.56 ± 2.06$^{\mathrm{a}}$	45.74 ± 5.51$^{\underline{\mathrm{a}}}$
	African Olive	41.50 ± 3.47$^{\mathrm{A}}$	43.08 ± 1.38$^{\mathrm{a}}$	43.53 ± 1.50$^{\underline{\mathrm{a}}}$
	Mix	44.76 ± 3.17$^{\mathrm{B}}$	45.61 ± 1.61$^{\mathrm{a}}$	43.53 ± 1.93$^{\underline{\mathrm{a}}}$
LI	Lucerne	42.24 ± 3.41$^{\mathrm{a}}$	46.21 ± 1.71$^{\mathrm{A}}$	41.93 ± 7.92$^{\underline{\mathrm{a}}}$
	Eucalyptus	44.52 ± 3.36$^{\mathrm{a}}$	49.40 ± 0.88$^{\mathrm{B}}$	46.16 ± 4.87$^{\underline{\mathrm{a}}}$
	African Olive	40.13 ± 2.73$^{\mathrm{a}}$	42.76 ± 1.22$^{\mathrm{C}}$	44.40 ± 2.70$^{\underline{\mathrm{a}}}$
	Mix	43.75 ± 3.11$^{\mathrm{a}}$	44.98 ± 1.41$^{\mathrm{AC}}$	43.52 ± 1.92$^{\underline{\mathrm{a}}}$

At time 0 there was an overall significant difference in total N (Table 2.10) of treatments in LI and II (one-way ANOVA, Tukey's HSD test; $P < 0.001$ for both) but not in CPW ($P = 0.415$). Overall, after 6 months of incubation, total N of Eucalyptus litter was greater than all other treatments (one-way ANOVA, Tukey's HSD test; $P < 0.05$) and independent of vegetation type. The African Olive treatment tended to have the smallest amount of N in all vegetation types.

The decomposition bags retrieved from the LI and II areas after 12 months of incubation showed a similar pattern among treatments to that found after 6 months with an overall enrichment in N. This enrichment was not evident for CPW and the Eucalyptus treatment still had the highest total N (one-way ANOVA, Tukey's HSD; $P < 0.05$). After 17 months of incubation, total N of all decomposed plant material ranged between 1.8 and 2.2% across all treatments and vegetation types and no statistical differences were found (one-way ANOVA, Tukey's HSD test; $P > 0.05$).

Table 2.10. Nitrogen content (%) of four different decomposition treatments (Lucerne, Eucalyptus, African Olive and Mix) after 6, 12 and 17 months of incubation for three vegetation types (Cumberland Plain Woodland (CPW), initial invasion (II) and long-term African Olive invasion (LI)) at the Australian Botanical Garden, Mount Annan. Different letters represent significant statistical differences. The comparisons were made within one vegetation type at each time between treatments.

		Incubation time (months)		
Vegetation type	Treatment	6	12	17
CPW	Lucerne	$1.68 \pm 0.34^{\mathrm{ab}}$	$1.79 \pm 0.27^{\underline{\mathrm{a}}}$	2.21 ± 0.45^{a}
	Eucalypt	$1.94 \pm 0.17^{\mathrm{a}}$	$2.07 \pm 0.14^{\underline{\mathrm{a}}}$	1.88 ± 0.17^{b}
	Olive	$1.19 \pm 0.67^{\mathrm{ab}}$	$1.81 \pm 0.04^{\underline{\mathrm{ab}}}$	2.00 ± 0.23^{a}
	Mix	$1.76 \pm 0.18^{\mathrm{ab}}$	$2.04 \pm 0.06^{\underline{\mathrm{ab}}}$	1.97 ± 0.09^{a}
II	Lucerne	$1.66 \pm 0.35^{\mathrm{AB}}$	$1.70 \pm 0.16^{\mathrm{a}}$	$2.20 \pm 0.44^{\underline{\mathrm{a}}}$
	Eucalypt	$1.83 \pm 0.25^{\mathrm{B}}$	$2.10 \pm 0.13^{\mathrm{a}}$	$1.85 \pm 0.12^{\underline{\mathrm{a}}}$
	Olive	$1.49 \pm 0.07^{\mathrm{A}}$	$1.81 \pm 0.04^{\mathrm{a}}$	$1.91 \pm 0.08^{\underline{\mathrm{a}}}$
	Mix	$1.72 \pm 0.16^{\mathrm{B}}$	$2.01 \pm 0.12^{\mathrm{a}}$	$1.97 \pm 0.09^{\underline{\mathrm{a}}}$
LI	Lucerne	$1.54 \pm 0.16^{\mathrm{ab}}$	$1.66 \pm 0.12^{\mathrm{A}}$	$2.16 \pm 0.43^{\underline{\mathrm{a}}}$
	Eucalypt	$1.76 \pm 0.20^{\mathrm{b}}$	$2.02 \pm 0.09^{\mathrm{B}}$	$1.91 \pm 0.17^{\underline{\mathrm{a}}}$
	Olive	$1.45 \pm 0.07^{\mathrm{a}}$	$1.84 \pm 0.06^{\mathrm{AB}}$	$1.99 \pm 0.11^{\underline{\mathrm{a}}}$
	Mix	$1.69 \pm 0.14^{\mathrm{ab}}$	$1.96 \pm 0.13^{\mathrm{B}}$	$2.01 \pm 0.17^{\underline{\mathrm{a}}}$

Change in C:N ratio in litter

The changes in total C and N can be integrated and expressed as changes in C:N ratio (Table 2.11). Overall, there was a significant effect of incubation time on C:N ratio for LI (RM ANOVA, Tukey's HSD test; $F = 968.1$, $P < 0.001$), II ($F = 221.9$, $P < 0.001$) and CPW areas ($F = 13.8$, $P = 0.002$). After 6 months of incubation, the treatments in II showed statistically significant differences with the African Olive treatment having a higher C:N ratio than the Eucalyptus (one-way ANOVA, Tukey's HSD test; $P = 0.039$) and Mix treatments (one-way ANOVA, Tukey's HSD test; $P = 0.041$). No differences were found between treatments with African Olive and Lucerne (one-way ANOVA, Tukey's HSD test; $P = 0.224$), Mix and Lucerne ($P = 0.735$) and Eucalyptus and Lucerne ($P = 0.724$). Overall in LI, the Lucerne treatment was statistically different from the other treatments after 12 and 17 months of incubation in II and LI (one-way ANOVA, Tukey's HSD test; $P < 0.05$).

Table 2.11. C:N ratio of four decomposition treatments (Lucerne, Eucalyptus, African Olive and Mix) after 6, 12 and 17 months of incubation for three vegetation types (Cumberland Plain Woodland (CPW), initial invasion (II) and long-term African Olive invasion (LI)) at the Australian Botanical Garden, Mount Annan. Different letters represent significant statistical differences within one vegetation type at each time among treatments.

		Incubation time (months)		
Vegetation type	Treatment	6	12	17
CPW	Lucerne	26.01 ± 3.40^{a}	26.25 ± 4.05$^{\underline{a}}$	18.99 ± 0.91$^{\mathit{a}}$
	Eucalypt	24.85 ± 0.97^{a}	23.6 ± 2.52$^{\underline{a}}$	24.41 ± 1.23$^{\mathit{a}}$
	Olive	27.94 ± 2.03^{a}	23.65 ± 0.75$^{\underline{a}}$	21.89 ± 2.34$^{\mathit{a}}$
	Mix	25.85 ± 3.42^{a}	21.86 ± 0.76$^{\underline{a}}$	22.27 ± 0.86$^{\mathit{a}}$
II	Lucerne	26.39 ± 3.60^{AB}	27.42 ± 2.32^{a}	19.04 ± 0.90$^{\underline{a}}$
	Eucalypt	25.27 ± 1.43^{B}	23.29 ± 2.41^{a}	24.66 ± 1.35$^{\underline{a}}$
	Olive	27.94 ± 2.03^{B}	23.74 ± 0.77^{a}	22.82 ± 1.42$^{\underline{a}}$
	Mix	26.15 ± 3.34^{B}	22.74 ± 1.63^{a}	22.11 ± 0.58$^{\underline{a}}$
LI	Lucerne	27.57 ± 1.69^{a}	27.90 ± 1.83^{A}	19.51 ± 1.47$^{\underline{a}}$
	Eucalypt	25.40 ± 1.32^{a}	24.55 ± 1.40^{B}	24.20 ± 1.56$^{\underline{b}}$
	Olive	27.66 ± 1.86^{a}	23.29 ± 0.89^{B}	22.29 ± 0.89$^{\underline{b}}$
	Mix	26.10 ± 3.36^{a}	23.06 ± 1.71^{B}	21.75 ± 1.12$^{\underline{ab}}$

Decomposition constant

The decomposition constant k (years^{-1}) was calculated for each treatment within each fuel type (Table 2.12). There was no significant difference (one-way ANOVA, Tukey's HSD test; P >0.05) of k values among different vegetation types indicating that regardless of where the decomposition bags were placed, the rates of decomposition were similar. The only statistical difference found occurred for treatments in II where Lucerne had a greater k value than Eucalyptus (one-way ANOVA, Tukey's HSD test; P = 0.002), African Olive (P = 0.003) and Mix (P = 0.006) treatments.

Table 2.12. Decay constant (k year^{-1}) values for four different decomposition treatments (Lucerne, Eucalyptus, African Olive and Mix) for three vegetation types (Cumberland Plain Woodland (CPW), initial invasion (II) and long-term African Olive invasion (LI)) at the Australian Botanical Garden, Mount Annan. Different letters represent significant statistical differences. The comparisons were made within one vegetation type at each time among treatments.

Treatment	Vegetation type		
	LI	II	CPW
Lucerne	0.91 ± 0.24$^{\mathrm{a}}$	1.14 ± 0.09$^{\mathrm{a}}$	1.09 ± 0.16$^{\mathrm{a}}$
Eucalypt	0.96 ± 0.45$^{\mathrm{A}}$	0.60 ± 0.14$^{\mathrm{A}}$	0.79 ± 0.17$^{\mathrm{A}}$
Olive	0.59 ± 0.18^{a}	0.66 ± 0.11^{a}	0.75 ± 0.13^{a}
Mix	0.80 ± 0.11$^{\underline{\mathrm{a}}}$	0.70 ± 0.11$^{\underline{\mathrm{a}}}$	1.02 ± 0.36$^{\underline{\mathrm{a}}}$

2.3.6. Vegetation type flammability

The four flammability-related measures are presented in Figure 2.5. Fuel from II ignited twice as fast as fuel from LI (one-way ANOVA, Tukey's HSD test; P <0.001), and moderately faster than fuel from CPW (P = 0.009). Flame duration for fuel from II was shorter than for fuel from CPW (one-way ANOVA, Tukey's HSD test; P = 0.032) and LI (P <0.001). Fuel from II had a greater amount of unburned mass remaining after combustion (residual mass fraction) when compared to LI (one-way ANOVA, Tukey's HSD test; P = 0.028) but no difference was found when compared to CPW (P = 0.094).

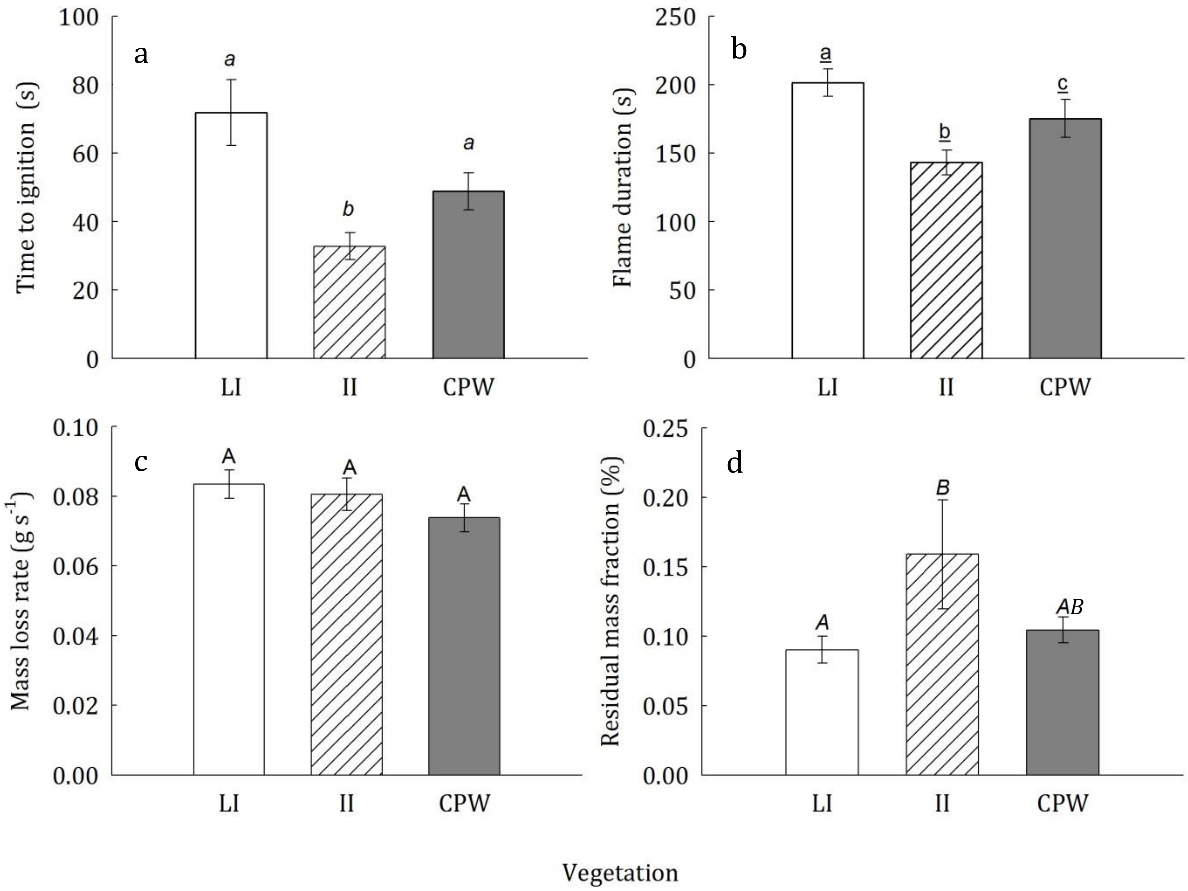

Figure 2. 5. Flammability measures including (a) time to ignition (s), (b) flame duration (s), (c) mass loss rate (g s-1), and (d) residual mass fraction (%) for three vegetation types (Cumberland Plain Woodland (CPW), initial invasion (II) and long-term African Olive invasion (LI)) at the Australian Botanical Garden, Mount Annan. Different letters represent significant statistical differences among vegetation types. Bars represent mean values and error bars are standard error.

2.4. Discussion

2.4.1. Vegetation vertical structure

In general, plants show considerable variation in their structure varying in response to the environmental conditions therefore allowing architectural rearrangements (Rowe and Speck 2005). This variation can potentially affect fire behaviour. In addition, invasive plants can significantly change the distribution of available fuels (Addo-Fordjour *et al.* 2009; Berry *et al.* 2011) and may alter fire behaviour (Mack and D'Antonio 1998; Brooks *et al.* 2004; Walther *et al.* 2009). Analysis of the vertical structure of vegetation can provide insight to how fire may ultimately behave in a given fuel type (Graham *et al.* 2004; Fernandes 2009).

It was clear that the vertical structure of the three vegetation types chosen for this study differed. Areas that had long-term invasion by Africa Olive were structurally less complex than Cumberland Plain Woodland and areas of intermediate invasion. Such differences in vertical arrangement of fuel could have a profound influence on the development and propagation of fire. For example, if fuels are discontinuous between surface and elevated layers, there is a smaller chance that fire will move up into the canopy (Cruz *et al.* 2010; Cruz *et al.* 2013). Under normal circumstances, the fuel arrangement, configuration and orientation within the fuel strata is complex and dynamic (Walker 1981; Pyne 1984). However, plant invasions can shift the fuel structure to a new climax vegetation (Mandle *et al.* 2011; Lockwood *et al.* 2013) and permanently alter the vegetation structure.

Cuneo and Leishman (2012) showed that African Olive is capable of forming a dense and permanent mid-canopy in grassy woodland vegetation forming a shadow that obstructs the light and ultimately kills the ground vegetation and does not allow growth of smaller plants. Although there are many ecological effects involved in structural alteration of vegetation, this is the first study showing how the fine fuel is vertically distributed in patches densely invaded by this species.

Areas with a well-established canopy of African Olive had very little fuel vertically distributed from 0 to 50 cm creating a gap between the surface fuel and the top layers. The opposite was found in Cumberland Plain Woodland and areas of intermediate invasion where there was much greater vertical continuity of fuel. The vertical complexity found in Cumberland Plain Woodland is mostly due to the presence of a characteristic grass layer with occasional shrubs spread throughout the understorey (Benson and Howell 2002). Areas of intermediate invasion had the greatest vertical complexity of all the vegetation types examined due to the presence of young individuals of African Olive in a matrix of native shrubs and a tall grass layer. Vertical continuity of fuel is important for fire behaviour as it can create a ladder for the fire to propagate to elevated fuel and the canopy (Whelan 1995). Even though there was considerable vertical fuel continuity in areas of intermediate invasion, it is important to note that the horizontal distribution of shrubs and young individuals of African Olive was patchy due to previous land use. These areas also lacked taller eucalypt trees and therefore there was no distinct canopy as in Cumberland Plain Woodland. Consequently, fire behaviour expected in this vegetation type would be more similar to a grassfire than a forest fire.

It is also important to note that most of the fuel above 50 cm in areas of intermediate invasion and long-term invasion was composed of live twigs and leaves that require pre-heating and moisture loss before igniting. In contrast, in areas of long-term invasion by African Olive, the presence of a considerable amount of dead fine fuel above 50 cm gives this vegetation the right conditions for a fire to propagate upwards into the canopy but a lack of surface and near surface are likely to impose a barrier for vertical fire propagation.

2.4.2. Fine fuel load

The importance of the fuel load and distribution in determining fire behaviour in different vegetation types is extremely variable. The time taken to build up levels of available fuel after a fire event and the condition of the fuel dictates the fire behaviour in that area (Whelan 1995).

Fuel arrangement and distribution can sometimes be more important in determining fire behaviour than fuel loads (Gould *et al.* 1997; Smith *et al.* 2004). The processes of describing and quantifying fuels is highly important for understanding fire behaviour and can provide information for fire management activities including prescribed burning, suppression difficulty, fuel hazard assessment and fuel treatment (Gould *et al.* (2011). However, the structure of fuel needs to be analysed together with fuel load to provide an understanding of how a fire can spread through the vegetation. For example, the presence of fuel in the upper layers in Cumberland Plain Woodland and areas invaded by African Olive does not necessarily mean that the fuel load is sufficient to carry a fire.

Fuel loads are important inputs for fire behaviour modelling and management of planned fires (Hessburg *et al.* 2007) and for wildfire management (Black and Opperman 2005). Information about the fuel load can also show dead and live carbon storage pools and indicate the potential of ecosystems as possible carbon sinks (Finkral and Evans 2008). Keane (2013) reviewed the challenges of describing fuels focusing discussions on surface fuel loadings as the primary characteristic used by fire scientists.

Until very recently fuel load was the only characteristic used in Australian fire danger systems to predict fire behaviour in forests (Gould *et al.* 2007a). Fuel reduction practices in eucalypt forest in Australia have relied on the relationships among fuel load, rate of spread and fire intensity as its foundation for over 30 years (Gould *et al.* 2007a). These relationships were proved to be weak under high intensity fires (Cheney 1990; Burrows 1994; Cheney and Gould 1996; Burrows 1999) and it has been suggested that variables other than fuel load could be better predictors of fire behaviour in forests (McCaw *et al*. 2008).

The ratio of dead and live fine fuel constitute an important characteristic capable of explaining sustained fire propagation in the elevated layer (Cruz *et al.* 2010). There was no difference in total fine fuel load amongst the three vegetation types and most of the fuel was found in the litter layer and was dead (more than 80%). The surface fuel loads measured in this study (0.33–0.72 kg m^{-2}) were within the same range of the surface fuel loads found by Gould *et al.* (2011) for open dry eucalypt forest (*Eucalyptus marginata*) in Western Australia (from 0.45–1.19 kg m^{-2}), however, the near surface (0–50 cm) fuel load of CPW (0.04 kg m^{-2}) was smaller compared to other studies in eucalypt forests (Gould *et al.* 2011; Thomas *et al.* 2014). Sites with long-term invasion of African Olive had low fuel loads in the upper canopy layers as a consequence of the structure of mature trees. In contrast, Cumberland Plain Woodland had low fuel load above the litter layer possibly as a consequence of the past history of prescribed burning or other disturbance such as grazing or due to low productivity of this vegetation type in general.

Areas of intermediate invasion were mostly grass with a few sparse shrubs and young African Olive trees. The structure and fuel load more closely resembled grassland. For a long time, fuel load was believed to be one of the most important variables to predict fire intensity and rate of spread in this vegetation type. In Australia, the first models developed to predict rate of spread in grassland used to incorporate fuel load as a direct input (McArthur 1966). However, subsequent research by Cheney *et al.* (1993; 1998) found that the role of fuel load in defining the rate of spread was minimal. Instead of fuel load, the condition of the grass (grazed or non-grazed) was a much more important factor in determining rate of spread in grassland.

At the moment, models available for predicting fire behaviour in eucalypt forests without using fuel load as the main variable are still being developed and tested (Gould *et al.* 2011; Watson *et al.* 2012; Cruz *et al.* 2013). Fuel load remains one of the most important inputs for prediction of fire behaviour given the current lack of specific models for predicting fires in areas invaded by woody weeds.

2.4.3. Fuel hazard score and percentage cover score

The overall fuel hazard score (FHS) and percentage cover score (PCS) of heavily invaded areas was greater than for areas of intermediate invasion and Cumberland Plain Woodland with the exception of the near-surface layer. As discussed previously, areas of long-term invasion by African Olive had a ‘gap’ in lower fuel layers which was also reflected in the low values recorded for FHS and PCS. As there are no other studies that have used hazard assessment methodologies for describing a novel fuel type, current methods were adapted in this study. To do this, fuels in areas of intermediate invasion and Cumberland Plain Woodland were measured using the same layers created to assess fuel hazard in eucalypt forests and the PCS and FHS were determined using tables in Gould *et al.* (2011). The assessment of areas of long-term invasion by African Olive followed the same scheme described by Gould *et al.* (2011), with adaptations for novel vegetation to conform to the same classes created for eucalypt forests.

When the FHS obtained for the surface and near surface fuels are combined they can be used to predict the rate of spread in vegetation using the tables provided in Gould *et al.* (Gould *et al.* 2007b). However, these tables were developed for specific conditions and forest type so using them to predict fire behaviour in invaded areas could produce misleading results (see Chapter 5 for discussion of fire behaviour prediction).

Watson *et al.* (2012) noted that although FHS were not designed for the purpose of measuring or implying fuel load they have been widely used by fire managers in Australia with this purpose and to make fire behaviour predictions using the McArthur models (McArthur 1967; Noble *et al.* 1980). The use of FHS to predict fuel load of an area can be extremely inaccurate leading to wrong fire behaviour predictions (Watson *et al.* (2012). Therefore, due to inherent inaccuracies, it cannot be recommended that FHS and PCS should be used to predict the fuel load and fire behaviour of invaded sites. Similarly, using Vesta fire behaviour tables to predict fire behaviour in areas invaded by African Olive is likely to lead poor predictions because Vesta models are empirical and specifically tailored for *Eucalyptus* forests. This could have a considerable impact on management actions.

Nevertheless, because FHS and PCS influence fire behaviour and the FHS, PCS and fuel depth of LI areas is highly contrasting when compared to II and CPW areas it is likely that the fire behaviour in the invaded patches would be different. The extent of the effect of these differences is extremely hard to predict without the observation of real fires and the development of empirical models. Watson *et al.* (2012) indicate that even though collection of fuel load measurements is time consuming these data are much more reliable than estimates derived from FHS. This would be particularly important when the vegetation being considered is a patch of forest invaded by a non-native species.

The FHS and PCS obtained for the invaded areas could potentially be used to compare and calibrate fire behaviour predictions made by physical or quasi-physical models capable of using the fuel load/depth and structure measurements as inputs to make fire behaviour predictions. As shown by Watson *et al.* (2012), the development of the FHS and PCS concept created a useful option to assess fuel condition and can be applied to many different types of vegetation. However, using it in fire management decisions is imprudent especially when considering vegetation subject to plant invasion.

2.4.4. Litter decomposition and soil

Decomposition of leaf litter has a key role in nutrient cycling in terrestrial ecosystems, as it is the main source of nutrients and organic matter for plant roots and soil organisms (Ashton *et al.* 2005). From a fuel perspective, litter decomposition can be an indication of how much litter a vegetation can potentially accumulate over time. Generally, leaf litter with a high N concentration decomposes before other litter types due to the lower energetic costs involved in breaking it down (Melillo *et al.* 1982).

Ashton *et al.* (2005) showed that interspecific differences in leaf litter quality could affect rates of decomposition, which can feedback to soil processes. The surface soils samples (0–5 cm) used in this study did not show any differences in N and C content and pH among vegetation types. The deeper soil layer (5–10 cm) of partially and fully invaded areas had higher concentrations of N and C than in Cumberland Plain Woodland. This could be due to invasion of African Olive altering soil processes in some way. Invasive species generally have a higher concentration of leaf N (Vitousek and Walker 1989; Witkowski 1991) and consequently decompose faster than native plants, introducing more N to the soil (Ashton *et al.* 2005). This pattern was evident with treatments using Lucerne which had the greatest N concentration and mass decay after 6 months of decomposition. In contrast, litter from African Olive had the smallest N concentration and the slowest decomposition rates of all treatments used. Low levels of N in fallen leaves of African Olive could be due to several factors including reallocation of nutrients before leaves are shed, reallocation to support fruit growth (Fernandez-Escobar *et al.* 2004), limited availability of N in the environment (Aerts 1996; Toberman *et al.* 2014) or leaching of soluble N from freshly fallen leaves. The N content of litter is also important for assimilation by microorganisms decomposing the material and C:N ratio has been shown to influence litter biodegradation, rates of mineralization and microbial biomass (Singh *et al.* 2014). Based on this information, slow rates of decomposition would be expected for litter with a high C:N ratio, similar to African Olive. This in turn is likely to promote accumulation of surface fuel which will have implications for fire behaviour in areas heavily invaded by woody weeds.

Differences in initial decomposition rates were expected during the first 6 months due to the differences found in their initial C:N ratio (Berg *et al.* 2003), however, initial C:N ratio alone does not accurately predict rates of decomposition (O'Connell and Menage 1983; Simpson 2005; Gul *et al.* 2012; Norris *et al.* 2013). These differences can also be attributed to lignin content that was not measured in this work (Melillo *et al.* 1982; Berendse *et al.* 1987).

Even though differences in rates of decomposition were expected amongst vegetation types only areas of intermediate invasion showed treatment differences. This may be due to these areas being more open than areas of long-term invasion and Cumberland Plain Woodland. The decomposition of forest litter can be influenced by climate conditions (Florence and Lamb 1975), aspect, species and slope position (Bale and Charley 1994; Mudrick *et al.* 1994), litter supply (Nakane 1995; Thomas *et al.* 2014), acidity (Berger and Glatzel 1996) and soil fertility (Klemmedson 1992).

The type of litter can influence physical mass loss and chemical changes while the decomposition process takes place (Singh and Gupta 1977; Aber *et al.* 1990; Francesca Cotrufo *et al.* 1995; Simpson 2005). The quality and quantity of litter changes through the decomposition process and the microbial community is also altered (Berg and McClaugherty 2003; Norris *et al.* 2013). Therefore, interactions between decomposers and the chemical composition of litter controls decomposition at different stages of decay (Rinkes *et al.* 2014). Eucalypt species can take from 7 to 375 years to lose 95% of their mass (Mackensen *et al.* 2003). Although the leaves of eucalypt used in this study had a relatively high N content and were expected to decay faster than leaves from African Olive, decomposition rates did not differ.

Thomas *et al.* (2014) modelled rates of decomposition (k year^{-1}) for dry sclerophyll forests in south-eastern Australia and calculated that they varied between 0 and 0.7 k year^{-1} for areas with 600 mm of rain. In this study, rates of decomposition for Cumberland Plain Woodland were slightly higher but this may be accounted for by higher annual rainfall at Mount Annan (approximately 800 mm). Rodriguez *et al.* (2009) measured a decomposition rate of 1.18 k year^{-1} for *Olea europea* cv *sylvestris* with 32% of the initial biomass remaining after 15 months. In this study, the decomposition rate for *Olea europea* var. *cuspidata* varied from 0.5–0.7 k year^{-1} and the remaining biomass after 17 months ranged from 30–35%. Decomposition rates can be closely related to precipitation and soil moisture and are strongly related to geographic factors (Zhang *et al.* 2008).

2.4.4. Vegetation type and flammability

The study of plant flammability is still relatively unexplored. Flammability was defined by Anderson (1970) and Martin *et al.* (1994) as having four components that can be related to fire characteristics at an ecosystem level (see Chapter 4).

Overall, fuel from the area of intermediate invasion had a faster time to ignition and shortest flame duration when compared to fuels from other vegetation types. Areas of intermediate invasion also had the greatest percentage of biomass left unburned or incompletely burned. This is most likely due to the type of vegetation in this area as it is dominated by grasses. Grass leaf blades and stems tend to be very light and the chemical composition is considerably different compared to leaves and twigs from woody vegetation (Brooks *et al*. 2004). These characteristics facilitate the ignition process and promote fast combustion of grasses compared to woody plants. The behaviour of grassfires can be demonstrated during controlled pyrolysis of grass material (Cheney and Sullivan 2008; Sullivan 2013). Consequently, grassfires tend to represent the greatest challenge in terms of fire suppression due to fast rates of spread and being highly influenced by wind and moisture (Cheney and Sullivan 2008). According to Sullivan (2003), grass fuels are characterised by a relative fineness when compared to coarser forest litter fuels and will ignite and burn faster for a given set of conditions with an average flaming time of 5 s and burning out in 10–15 s. It is likely that longer flaming times and higher residual mass fraction found in this study were due to the arrangement and compactness of the fuel when prepared for combustion in the MLC. The leaves and stems were trimmed to fit the holder, the material was arranged in a more compact way (i.e. different bulk density) and was positioned horizontally. In the field, grass fuel is arranged vertically with enough air between leaves to facilitate faster and more complete combustion.

The only differences found between flammability of fuel from long-term invasion sites and Cumberland Plain Woodland was flame duration. This difference could be due to the physical composition of the fuel. The fuel from African Olive had a greater number of thin twigs than Cumberland Plain Woodland which was mostly composed of leaves. The surface area:volume ratio of twigs is generally smaller than for leaves which could promote slower rates of combustion.

The flammability of the individual components of plants (i.e. leaves and twigs) can be extremely different from large-scale or 'ecosystem' flammability and there are many challenges to face when trying to scale up from the leaf-level to whole ecosystems (Gill and Zylstra 2005). Understanding the four components of flammability through laboratory experiments could help us to understand how ecosystem flammability works and could improve physical models of fire behaviour. With this in mind, an in-depth study of flammability is presented in Chapter 4.

3. Alteration of fuel load and structure by a native woody environmental weed, Cootamundra Wattle (*Acacia baileyana*)

3.1. Introduction

Species that are able to invade native vegetation and have the potential to permanently alter and destroy an ecosystem are often referred to as environmental weeds (Humphries *et al.* 1991; Carr *et al.* 1992; Morgan *et al.* 2002). These species can be considered to be one of the greatest threats to the preservation of natural environments in Australia and New Zealand (Humphries *et al.* 1991; Williams and West 2000). Environmental weeds do not necessarily need to be introduced species from another country (Morgan *et al.* 2002); native species that have extended their range beyond their natural distribution can also become environmental weeds (Williams and West 2000).

In Australia, environmental weeds that are also native species have been associated with the extinction of four plant species and threaten several more (Groves and Willis 1999). Examples of native woody weeds growing outside of their normal distribution in Australia include Sweet Pittosporum (*Pittosporum undulatum*; Mullett and Simmons 1995; Rose and Fairweather 1997) in Victoria and New South Wales (NSW), Golden Wreath Wattle (*Acacia saligna*) in Western Australia (Emms *et al*. 2005) and Coastal Tea Tree (*Leptospermum laevigatum)* in north-east NSW, south-east Queensland and Western Australia (Groves *et al.* 2005).

Williams and West (2000) classify environmental weeds as a subset of invasive plants because the problems caused by these species cannot be simply classified in economic or agronomic terms. Environmental weeds represent a particular challenge for land managers due to their effects on ecosystem stability, functional complexity and biodiversity (Adair and Groves 1998). In Australia, a native environmental weed that is becoming increasingly more important is *Acacia baileyana*, commonly known as Cootamundra Wattle. Cootamundra Wattle grows naturally in open woodlands (e.g. mallee communities). These plants form shrubs or small trees with a spreading crown, usually growing from 3 to 6 m tall. The natural distribution of this species is restricted to inland parts of southern NSW however it has spread from gardens (Groves *et al.* 2005) and has increased its distribution range (Williams and West 2000). Cootamundra Wattle is considered to be a significant environmental weed in Victoria and the Australian Capital Territory (ACT) with emerging importance in south-eastern South Australia, south-western Western Australia, south-eastern Queensland, Tasmania and in many parts of NSW that are beyond its natural range, particularly in coastal districts and in the Blue Mountains region (Government of Queensland 2014).

Cootamundra Wattle has been cultivated since the 1800s suggesting that there has been a considerable time span for this species to spread into native ecosystems (Morgan *et al.* 2002). Cootamundra Wattle has two main colour forms (purple and green) and both are considered environmental weeds even though the purple variety is less aggressive (Morgan *et al.* 2002). Outside its natural range Cootamundra Wattle invades open woodland, heathland and grassland on a variety of soil types (Government of Queensland 2014). The invasiveness of Cootamundra Wattle is thought to be due to frequent fire activity stimulating mass seed germination (Smith 1993), short development time to reproductive maturity (<2 years) and production of a large number of viable seeds with considerable longevity (Morgan *et al.* 2002).

Unlike African Olive investigated in Chapter 2, Cootamundra Wattle is a native species that has evolved under the same environmental conditions of other Australian plants. It would therefore be reasonable to assume that the flammability of a native environmental weed would be similar to the native plants when it is invading a habitat similar to that in which it evolved.. This scenario would allow for an investigation of mechanisms other than flammability (e.g. fuel structure and load) that may be involved in changes in fire behaviour of invaded areas. To investigate how a native environmental weed can affect fire behaviour, the following questions were formulated:

1. What are the differences between areas invaded with Cootamundra Wattle and the surrounding native woodlands in terms of fine fuel load and structure?
2. Does an invasion by Cootamundra Wattle represent a higher fire hazard?
3. Is the flammability of areas invaded by Cootamundra Wattle different from native woodland and areas showing signs of initial invasion?

3.2. Materials and methods

3.2.1. Study site

Canberra Nature Park, ACT is composed of 33 separate areas with vegetation ranging from lowland native grasslands to remnant woodlands (Australian Capital Territory Government 2014). The Red Hill Nature Reserve is part of the Canberra Nature Park and occupies an area of 375 ha of Yellow Box-Red Gum Woodland. At the Federal level, this vegetation type is listed as a Critically Endangered vegetation community under the *Environmental Protection and Biodiversity Conservation Act* 1999 (EPBC Act 1999) and in the ACT, Yellow Box-Red Gum Grassy Woodland is listed as an Endangered community (Nature Conservation Act 1980). This vegetation type has high plant diversity and is habitat for a number of threatened plant species (Red Hill Bush Regeneration Group 2014). The average annual rainfall is 616 mm with mean annual maximum temperature of 19.7 °C and mean annual minimum of 6.5 °C (Figure 3.1) (Australian Bureau of Meteorology 2014). Most of the Red Hill Nature Reserve is composed of metamorphic rock with occasional Silurian volcanic rocks outcrops (Red Hill Bush Regeneration Group 2014). Yellow Box-Red Gum Woodland has been highly fragmented and generally exists as isolated patches smaller than 5 ha in area (Gibbons and Boak 2002).

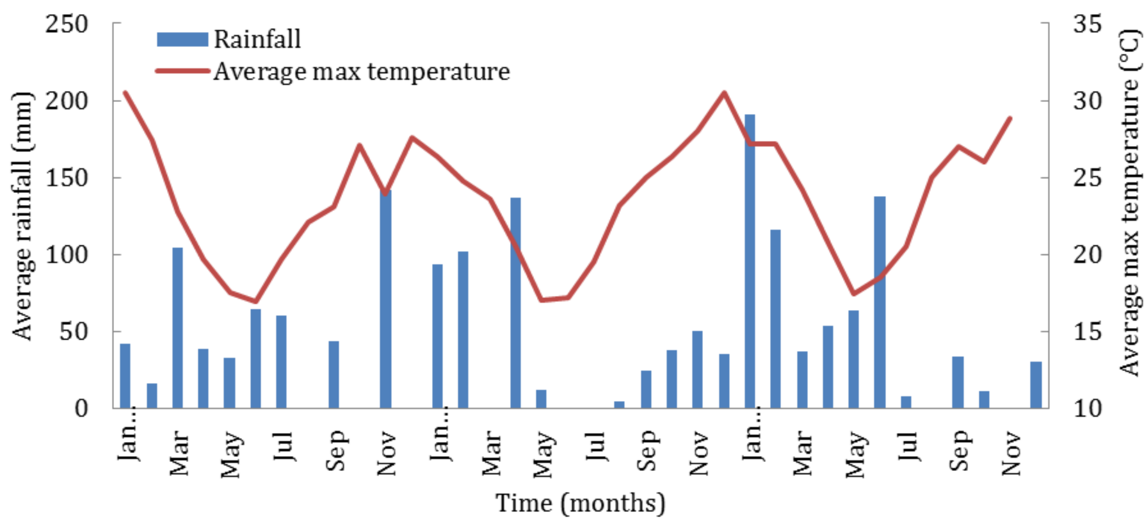

Figure 3. 1. Average rainfall and temperature for 2011, 2012 and 2013 for Canberra Airport weather station, ACT 35.31° S, 149.20° E (Data obtained from Bureau of Meteorology - http://www.bom.gov.au/climate/data/stations/).

The study area is located on lower slopes and gently undulating terrain, at an altitude of between 600–900 m. Because of the relatively large size of the Red Hill Nature

Reserve and that much of the understorey is in good condition, the reserve supports one of the highest native plant diversities recorded in a Yellow Box-Red Gum Woodland remnant anywhere in Australia with approximately 175 native species have been recorded in the Red Hill Nature Reserve (Red Hill Bush Regeneration Group 2014). Pell and Tidemann (1997) described the vegetation in this area as predominantly grassy open savannah with Blakely's Red Gum (*Eucalyptus blakelyi*), White Gum (*E. rossii)*, Yellow Box (*E. melliodora)*, Redbox Gum (*E. polyanthemos)* and Apple Box (*E. bridgesiana)* as the most common eucalypt species present in the area with a sparse and heterogeneous distribution ranging from isolated trees to areas of regrowth woodland. The understorey consists of a dense cover of native grasses and a scattered herbs and forbs, and with isolated native but non-eucalypt trees (e.g. *Acacia dealbata*, *Casuarina stricta* and *Exocarpus cupressiformis*). More disturbed areas of Yellow Box-Red Gum Woodland have introduced pasture grasses and shrubs (e.g. *Pyracantha* spp., *Rubus* spp. and *Rosa* spp.) but are still considered to be relatively intact.

The presence of Cootamundra Wattle in the Red Hill Nature Reserve represents a potential threat to the native vegetation since this species is declared as a C4 class under the *Noxious Weeds Act* (1993) (See Chapter 1) and its commercial and non-commercial supply is prohibited in the ACT.

3.2.2. Sampling design

The design of the study site was similar to that described in Chapter 2. Within the Red Hill Nature Reserve, two vegetation types were selected: well-preserved Yellow Box-

Red Gum Grassy Woodland (YGW; Figure 3.2a) and the same woodland showing initial stages of invasion characterised by sparsely distributed individuals (referred to as sparsely invaded areas; SI) of Cootamundra Wattle (Figure 3.2b). The area representing SI was carefully selected as it was difficult to delineate as the original vegetation persisted (e.g. native grasses and shrubs, sapling eucalypts) with the occasional presence of Cootamundra Wattle. The location chosen to represent heavy invasion (HI) by Cootamundra Wattle was less than a 1 km away from the Red Hill Nature Reserve at a site located between Groom, Carruthers and Kent Streets (-35.327715, 149.092144). Prior to invasion by Cootamundra Wattle, this area was occupied by Yellow Box-Red Gum Grassy Woodland but due to urban development of the area and land use change, the original vegetation was highly disturbed and few overstorey eucalypt trees remained. The understorey of HI areas was dominated by Cootamundra Wattle (Figure 3.2c).

In each of the three vegetation types, 50 × 50 m plots (n = 3) were established to investigate the fuel complex. Three parallel transects of 50 m were established in each plot at 5, 25 and 45 m along the perpendicular side of the plot. Scoring for the pin point intersect method (see Section 2.2.3) was done along each 50 m transect at 11 observation points located 5 m apart (Canfield 1941). To measure fuel hazard score, percentage cover score and fuel depth, five circles of 5 m radius were established in the same transects at 5, 15, 25, 35 and 45 m (see Section 2.2.3). Along each transect, two quadrats of 1 × 1 m were randomly located to assess fine fuel biomass (see Section 2.2.3).

3.2.3. Measurement of fuel

To describe the structure of the fuel layers and stratum cover and height, the same methodology as described in Chapter 2 was used (see Section 2.2.3). The total number of touches of vegetation was recorded for each layer (0–20, 20–50, 50–100, 100–200 cm) and total touches were calculated for each vegetation type.

Quantification of fuel load was done by destructive sampling following the same design and methodology described in Chapter 2 (see Section 2.2.3).

Visual estimation of the fuel hazard score (FHS) and percentage cover score (PCS) was made according to methods described in Chapter 2 (see Section 2.2.3). The FHS and PCS were categorically rated from 0 to 4 for each of the fuel layers. The depth of the litter and height of the near surface and elevated fuel layers were measured according to methods described by McCarthy *et al.* (1999).

Figure 3. 2. Details of the three vegetation types used: (a) Yellow Box-Red Gum Grassy Woodland (YGW), (b) an example of mature Cootamundra Wattle in a mown area close to sparsely invaded areas (SI), and (c) areas heavily invaded by Cootamundra Wattle (HI) at Red Hill Nature Reserve, ACT. Yellow arrows indicate individuals of Cootamundra Wattle.

3.2.3. Assessment of fuel flammability

To assess the flammability of components of fuel from the three vegetation types a mass-loss calorimeter was used (see Section 2.2.6). After determining the fuel moisture (see Section 2.2.3), biomass samples from each layer within the same vegetation type were bulked to form a composite sample. Three sub-samples of each composite sample were combusted forming a total of 15 samples per vegetation type. The weight of each sub-sample varied according to the fuel composing it. The samples were trimmed to fit the holder to uniformly cover the exposed surface area and sample thickness was maintained at 5 cm. Burns were done using an irradiance of 25 kW m^{-2} and a 10 kV spark igniter was used to provide piloted ignition.

Ignitability, sustainability, combustibility and consumability were assessed (see Section 2.2.6) allowing overall flammability comparisons between vegetation types.

3.2.4. Statistical analysis

The structure of the fuel in the three vegetation types (YGW, SI and HI) was compared using one-way ANOVA with Tukey's Honest Significant Difference (HSD) post-hoc tests. Prior to statistical analysis, the point intersect data was square root transformed. The pin point intersect data (i.e. the number of touches in each layer and the total number of touches amongst vegetation types) were compared for each fuel class (live: grass, twigs/leaves and herbs, and dead fine fuel) amongst vegetation types for each individual layers separately using one-way ANOVA and Tukey's HSD Tests. Values for fine fuel biomass collected during destructive sampling were log-transformed to normalise the data prior to one-way ANOVA. Fuel depth and height were compared amongst vegetation types within each layer using one-way ANOVA combined with Tukey's HSD tests.

Data for each flammability component (heat rate release (HRR), mass loss rate (MLR), time-to-ignition (TTI), flame duration (FD)) were compared by using one-way ANOVA with Tukey's HSD analysis. To meet the criteria for using one-way ANOVA, the values of TTI and FD were log-transformed and values for MLR were arc sine-transformed. Values derived from the visual scoring system (i.e. overall FHS, and FHS and PCS for the five fuel layers: overstorey tree and canopy, intermediate tree and canopy, elevated, near surface and surface) were compared using one-way ANOVA combined with Tukey's HSD tests. All statistical analyses were made in the SPSS version 22.

3.3. Results

3.3.1. Fuel architecture

Despite the similar total number of touches, the vertical structure of HI was considerably different from SI and YGW. Yellow Box-Red Gum Grassy Woodland had only half the number of total touches than areas heavily invaded with Cootamundra Wattle (Table 3.1). The majority of touches in YGW (86%) was in the first 20 cm above ground and was mostly due to a layer of grass. The areas of SI had a structure similar to YGW with greater amount of touches for each fraction reflecting encroachment of the vegetation caused by the presence of Cootamundra Wattle. The total number of touches in HI and SI were significantly different from YGW (one-way ANOVA, Tukey's HSD test; $P < 0.001$). The areas heavily invaded with Cootamundra Wattle (HI) had a greater number of touches for every layer (one-way ANOVA, Tukey's HSD test; $P < 0.05$) in comparison to SI and YGW.

Table 3.1. Total number of touches (± standard deviation) in each fuel height layer (0–20, 20–50, 50–100, 100–150, 150–200 cm and total) for three different fuel types (Yellow Box-Red Gum Grassy Woodland (YGW), sparsely invaded areas (SI), and areas heavily invaded by Cootamundra Wattle (HI)) at Red Hill Nature Reserve, ACT. Statistical comparisons (one-way ANOVA and Tukey's HSD tests) were made among vegetation types within each layer. Different letters represent significant statistical differences.

	Vegetation type		
Fuel layer (cm)	YGW	SI	HI
150-200	8 ± 7^{a}	26 ± 2^{a}	76 ± 21^{b}
100-150	4 ± 8^{A}	33 ± 1^{A}	91 ± 27^{B}
50-100	3 ± 8$^{\underline{a}}$	26 ± 1$^{\underline{b}}$	87 ± 24$^{\underline{c}}$
20-50	15 ± 13^{a}	70 ± 4^{a}	91 ±15^{b}
0-20	185 ± 43^{AB}	239 ± 35^{B}	131 ± 35^{A}
Total	215 ± 28$^{\underline{a}}$	394 ± 23$^{\underline{b}}$	476 ±23$^{\underline{b}}$

The fuel composition followed the same pattern for each of the vegetation types. Most of the fuel was composed of fine live fuel (i.e. 69% in HI, 54% in SI and 78% in YGW) with far less dead fine fuel. Heavily invaded areas had significantly greater amounts of live fuel in every layer (one-way ANOVA, Tukey's HSD test; $P < 0.05$) when compared to YGW.

Table 3.2. Total number of touches (± standard deviation) of each class (live fine fuel (LFF) and dead fine fuel (DFF)) in each fuel layer (0–20, 20–50, 50–100, 100–150, 150–200 cm and total) for three different fuel types (Yellow Box-Red Gum Grassy Woodland (YGW), sparsely invaded areas (SI) and areas heavily invaded by Cootamundra Wattle (HI) at Red Hill Nature Reserve, ACT. Statistical comparisons (one-way ANOVA and Tukey's HSD tests) were made among vegetation types within each layer. Dashes indicate that no biomass was measured for this fuel layer. Different letters represent significant statistical differences.

	Vegetation type and fuel type					
	DFF			LFF		
Fuel layer (cm)	YGW	SI	HI	YGW	SI	HI
150-200	3 ± 11$^{\mathrm{a}}$	8 ± 3$^{\mathrm{a}}$	2 ± 4$^{\mathrm{a}}$	5 ± 9$^{\mathrm{A}}$	17 ± 4$^{\mathrm{A}}$	73 ± 19$^{\mathrm{B}}$
100-150	1 ± 19$^{\underline{\mathrm{a}}}$	14 ± 2$^{\underline{\mathrm{b}}}$	8 ± 8$^{\underline{\mathrm{ab}}}$	3 ± 11$^{\underline{\mathrm{A}}}$	20 ± 2$^{\underline{\mathrm{A}}}$	83 ± 13$^{\underline{\mathrm{B}}}$
50-100	2 ± 14^{a}	16 ± 2^{b}	19 ± 9^{b}	1 ± 6^{A}	9 ± 1^{A}	68 ± 8^{B}
20-50	3 ± 27$^{\mathrm{A}}$	35 ± 3$^{\mathrm{B}}$	37 ± 3$^{\mathrm{B}}$	12 ± 8^{a}	35 ± 2^{a}	54 ± 21^{b}
0-20	37 ± 59^{A}	106 ± 14^{B}	80 ± 8^{B}	147 ± 38$^{\underline{\mathrm{a}}}$	133 ± 36$^{\underline{\mathrm{b}}}$	51 ± 20$^{\underline{\mathrm{b}}}$
Total	46 ± 16$^{\underline{\mathrm{A}}}$	180 ± 16$^{\underline{\mathrm{B}}}$	147 ± 29$^{\underline{\mathrm{B}}}$	168 ± 21$^{\mathrm{a}}$	214 ± 20$^{\mathrm{b}}$	329 ± 16$^{\mathrm{b}}$

3.3.2. Fine fuel load

Overall, there were no significant differences (one-way ANOVA, Tukey's HSD test; $P > 0.05$) in total live or dead fine fuel load amongst vegetation types (Table 3.3). The largest proportion of the fine fuel was in the litter layer (64% in LI, 67% in IS and 80% in YGW). The presence of small amounts of live and dead fine fuel in the layers above 50 cm in HI and SI reflect the vertical structure indicated by the pin-point method in these areas (Table 3.3).

Table 3. 3. Fine fuel biomass (kg m^{-2}; mean ± standard deviation) in classes (live, dead and total) and fuel height layer (litter, 0–20, 20–50, 50–100, 100–150, 150–200 cm and total) for three vegetation types (Yellow Box-Red Gum Grassy Woodland (YGW), sparsely invaded areas (SI) and areas heavily invaded by Cootamundra Wattle (HI)) at Red Hill Nature Reserve, ACT. Statistical comparisons (one-way ANOVA and Tukey's HSD tests) were made among vegetation types within each layer. Different letters represent significant statistical differences among the logarithmic data and dashes indicate that no biomass was measured for this fuel layer.

	Fuel and vegetation type								
	Live fine fuel			Dead fine fuel			Total		
Fuel layer (cm)	YGW	SI	HI	YGW	SI	HI	YGW	SI	HI
150–200	-	0.02 ± 0.02^{a}	0.07 ± 0.05^{a}	-	-	-	-	0.01 ± 0.01$^{\underline{a}}$	0.02 ± 0.03$^{\underline{a}}$
100–150	0.01 ± 0.01^{a}	0.03 ± 0.03^{a}	0.04 ± 0.03^{a}	-	-	-	0.01 ± 0$^{\underline{a}}$	0.01 ± 0.01$^{\underline{a}}$	0.01 ± 0.02$^{\underline{a}}$
50–100	-	0.03 ± 0.03^{a}	0.06 ± 0.05^{a}	-	0.02 ± 0.02^{A}	0.01 ± 0.02^{A}		0.01 ± 0.02$^{\underline{a}}$	0.02 ± 0.02$^{\underline{a}}$
20–50	0.01 ± 0.01^{b}	0.02 ± 0.02^{ab}	0.06 ± 0.06^{a}	-	0.01 ± 0.01^{A}	0.03 ± 0.03^{A}	-	0.01 ± 0.01$^{\underline{a}}$	0.02 ± 0.02$^{\underline{a}}$
0–20	0.16 ± 0.12^{a}	0.12 ± 0.07^{a}	0.13 ± 0.12^{a}	0.04 ± 0.03^{A}	0.07 ± 0.04^{A}	0.13 ± 00.1^{A}	0.03 ± 0.03$^{\underline{a}}$	0.06 ± 0.03$^{\underline{a}}$	0.08 ± 0.07$^{\underline{a}}$
Litter	-	-	-	0.38 ± 0.34^{A}	0.53 ± 0.31^{A}	0.79 ± 0.31^{A}	0.33 ± 0.3$^{\underline{a}}$	0.39 ± 0.23$^{\underline{a}}$	0.55 ± 0.38$^{\underline{a}}$
Total	0.18 ± 0.05^{a}	0.22 ± 0.05^{a}	0.37 ± 0.07^{a}	0.42 ± 0.11^{A}	0.64 ± 0.15^{A}	0.96 ± 0.22^{A}	0.41 ± 0.06$^{\underline{a}}$	0.58 ± 0.07$^{\underline{a}}$	0.86 ± 00.1$^{\underline{a}}$

3.3.3. Visual scoring of fuel and fuel depth

The overall FHS of HI areas was statistically higher than SI for surface (one-way ANOVA, Tukey's HSD test; P = 0.012), near-surface (P <0.001) and elevated fuel (P = 0.006; Table 3.4). The same pattern was found between areas of SI and YGW for surface, near-surface and elevated fuel (one-way ANOVA, Tukey's HSD test; P <0.001 for all). Bark fuel did not differ amongst vegetation types. Areas of SI had statistically greater FHS for surface (one-way ANOVA, Tukey's HSD test; P = 0.009), near-surface and elevated fuel (P <0.001 for both) than YGW.

There were no statistical differences in PCS for areas of HI and SI for any fuel layer (Table 3.5). Heavily invaded areas had greater PCS for surface and elevated fuel than YGW (one-way ANOVA, Tukey's HSD test; P <0.001 for both). The same pattern was found for areas of SI with PS for surface (one-way ANOVA, Tukey's HSD test; P = 0.002) and elevated layers (P < 0.001) being greater than for YGW. The similarity between SI and HI may reflect the presence of the Cootamundra Wattle in both areas.

The fuel depth in areas of HI was greater than SI for surface (one-way ANOVA, Tukey's HSD test; P = 0.026) and near-surface fuel layers (P = 0.040). Heavily invaded areas also had greater fuel depth than YGW for surface, near-surface and elevated fuel layers (one-way ANOVA, Tukey's HSD test; P <0.001 for all). The near-surface layer in areas of SI and YGW was statistically smaller then HI (one-way ANOVA, Tukey's HSD test; P = 0.022).

Table 3. 4. Fuel hazard score (mean ± standard deviation) in each fuel layer (Surface, Near-surface, Elevated and Bark) for three different vegetation types (Yellow Box-Red Gum Grassy Woodland (YGW), sparsely invaded areas (SI) and areas heavily invaded by Cootamundra Wattle (HI)) at Red Hill Nature Reserve, ACT. Statistical comparisons (one-way ANOVA and Tukey's HSD tests) were made among vegetation types within the same fuel layer. Different letters represent significant statistical differences.

	Vegetation type		
Fuel layer	YGW	SI	HI
Surface	1.2 ± 0.7^{a}	1.8 ± 0.7^{b}	2.3 ± 1.0^{c}
Near-surface	1.3 ± 0.5^{A}	2.4 ± 0.6^{B}	3.0 ± 0.8^{C}
Elevated	1.1 ± 0.8^{a}	1.9 ± 0.8^{b}	2.5 ± 0.8^{c}
Bark	1.0 ± 1.1^{a}	1.5 ± 0.9^{a}	1.5 ± 1.7^{a}

Table 3.5. Percentage cover score (mean ± standard deviation) in each fuel layer (Surface, Near-surface, Elevated and Bark) for three different vegetation types (Yellow Box-Red Gum Grassy Woodland (YGW), sparsely invaded areas (SI) and areas heavily invaded by Cootamundra Wattle (HI)) at Red Hill Nature Reserve, ACT. Statistical comparisons (one-way ANOVA and Tukey's HSD tests) were made among vegetation types within the same fuel layer. Different letters represent significant statistical differences.

	Vegetation type		
Fuel layer	YGW	SI	HI
Surface	1.2 ± 0.6^{a}	1.8 ± 0.7^{b}	2.1 ± 0.7^{b}
Near-surface	2.5 ± 0.8^{A}	2.6 ± 0.9^{A}	2.6 ± 0.8^{A}
Elevated	1.1 ± 0.7^{a}	2.0 ± 0.9^{b}	2.4 ± 0.8^{b}
Canopy	0.9 ± 0.8^{a}	0.9 ± 0.7^{a}	1.0 ± 0.9^{a}

Table 3.6. Fuel depth (mean ± standard deviation) for different fuel layers (Surface, Near-surface, Elevated and Canopy) in three different vegetation types (Yellow Box-Red Gum Grassy Woodland (YGW), sparsely invaded areas (SI) and areas heavily invaded by Cootamundra Wattle (HI)) at Red Hill Nature Reserve, ACT. Statistical comparisons (one-way ANOVA and Tukey's HSD tests) were made among vegetation types within the same fuel layer. Different letters represent significant statistical differences.

	Vegetation type		
Fuel layer	YGW	SI	HI
Surface (mm)	10.2 ± 6.9^{a}	13.0 ± 6.5^{a}	16.7 ± 6.2^{b}
Near-surface (cm)	19.1 ± 6.7^{A}	24.4 ± 8.2^{B}	29.3 ± 1.2^{C}
Elevated (m)	1.0 ± 0.7^{a}	1.6 ± 0.4^{b}	1.8 ± 0.3^{b}

3.3.4. Vegetation type flammability

There were no statistical differences for any flammability component (ignitability, sustainability, combustibility or consumability) amongst the three vegetation types (Figure 3.2). Overall, TTI ranged from 25–45 s. Flame duration was sustained and ranged from 170–230 s. Although the results show no statistical differences for MLR, the fastest rate of mass loss was for fuel from HI areas and the slowest for fuel from YGW. As described in Chapter 2, this could simply reflect the packing ratio of the leaves in the holder prior to combustion. The RMF only varied between 16–18% for the three vegetation types.

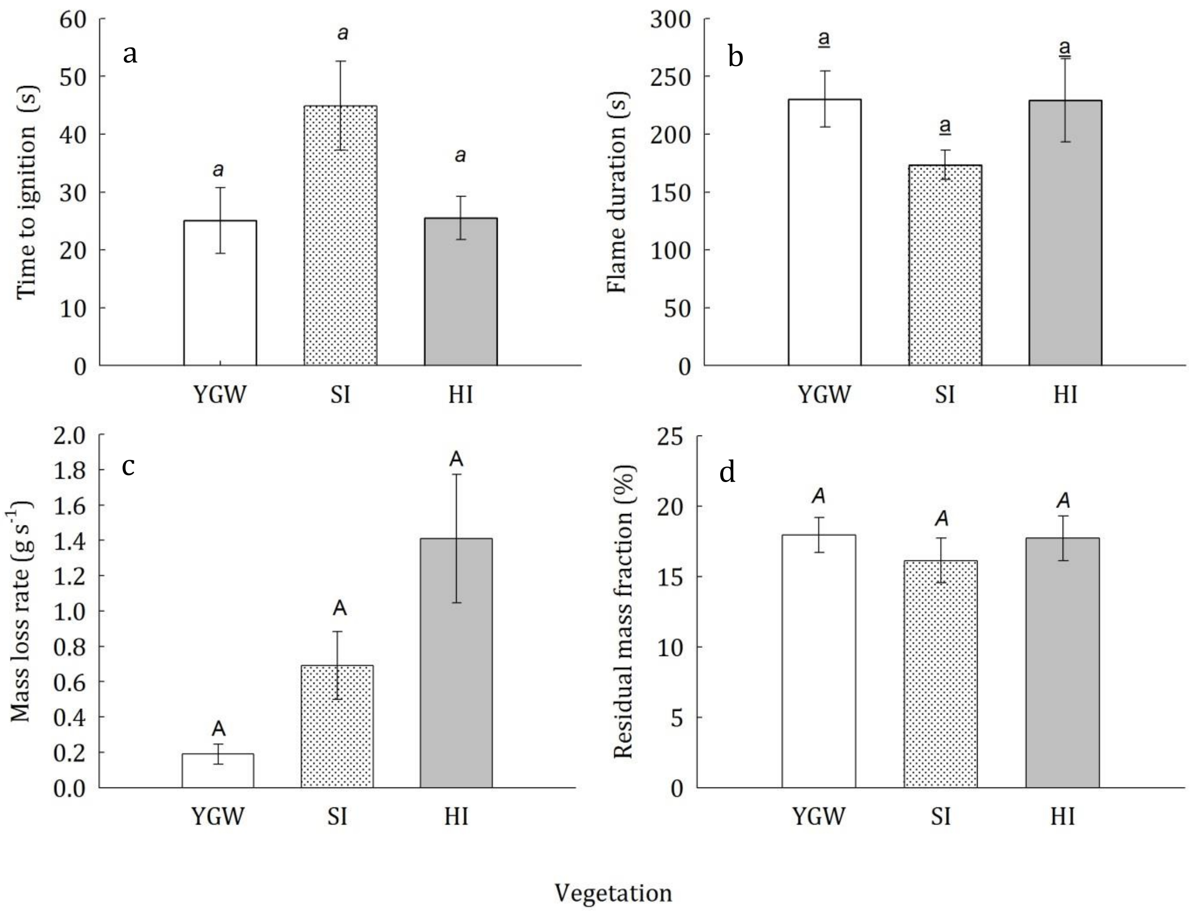

Figure 3. 3. Flammability measures including (a) time to ignition, (b) flame duration, (c) mass loss rate, (d) residual mass fraction of fuel from three different vegetation types (Yellow Box-Red Gum Grassy Woodland (YGW), sparsely invaded areas (SI) and areas heavily invaded by Cootamundra Wattle (HI)) at Red Hill Nature Reserve, ACT. Statistical comparisons (one-way ANOVA and Tukey's HSD tests, significance level of 0.05) were made among fuel types. Different letters represent significant statistical differences among vegetation types. Bars represent mean values and error bars are standard error.

3.4. Discussion

3.4.1. Alteration of the fuel and invasion by Cootamundra Wattle

The presence of a woody invasive species can change the fire behaviour or alter fire frequency through different mechanisms. Structural changes in the fuel are the most obvious modifications caused by the presence of invasive species (Pauchard *et al.* 2008; Berry *et al.* 2011). This study showed that Cootamundra Wattle altered the vertical structure of fine fuel in Yellow Box-Red Gum Grassy Woodland. In uninvaded areas, most of the fine fuel was in the 0–50 cm layer and was typically composed of native grasses with occasional herbs and forbs. Fine fuel was sparse above 50 cm height. In sparsely invaded areas there were only a few individuals of Cootamundra Wattle but their presence was enough to start to shift the vertical distribution of fine fuel towards the sub-canopy. In areas heavily invaded by Cootamundra Wattle there was a much greater distribution of fine fuel homogeneously spread from the ground up to 2 m.

The organisation and distribution of fuel can be more useful for defining fire behaviour than fuel loads (Gould *et al.* 1997; Smith *et al.* 2004). However, quantifying fuel is still extremely important (Gould *et al.* 2011) and fire prediction models currently used by Australian authorities still use fuel load as an input (Gould *et al.* 2007a). The total fine fuel load measured in this study (0.41–0.86 kg m^{-2}) is in the range of fine fuel load of grassy woodland predicted by the model developed by Thomas *et al.* (2014; 0.6–0.8 kg m^{-2}). Most of the fine fuel was found in the litter layer (surface layer). The surface fine fuel of woodland ecosystems is typically composed of litter originating from the woody plants and the herbaceous layer (Raison *et al.* 1983; Birk and Bridges 1989) and is considered to be one of the most important fuel layers due to its influence on the ignitability and rates of fire spread of a forest (Sullivan *et al.* 2012). Thomas *et al.* (2014) showed that the composition of the litter layer is strongly influenced by the leaf characteristics of the trees dominating that community. In *Eucalyptus* woodlands, the litter layer is expected to contain large amounts of relatively large leaves with high lignin content and therefore high flammability (Scarff and Westoby 2006). When the dominant species occurring in a community changes due to biological invasion, a change in the composition of the litter can be expected. The modification of the vertical fuel structure and the alteration of the leaves composing the litter layer are likely to be the main cause of any increase in fire hazard, frequency and behaviour in heavily invaded areas.

With the exception of the bark layer, the FHS corresponded to the degree of invasion by Cootamundra Wattle. Due to the alteration of the vertical distribution of the fuel in heavily invaded areas, the fuel hazard was expected to be higher. However, such a clear difference in FHS between the sparsely invaded areas and the woodland were not expected. The presence of Cootamundra Wattle also increased the overall PCS and depth of the different fuel layers. As shown for African Olive, the invasion of native vegetation can alter the fuel load and architecture. The invasion process may be different for each invasive species and the consequences for the invaded ecosystems vary widely (Mandle *et al.* 2011). Similarly, the way in which the invasive species affects the fire regimes will vary.

In this study, the flammability of fuel collected from the three different vegetation types did not vary. Plant species from different regions experience different fire regimes and exhibit adaptive responses to the direct effects of the fire (Schwilk and Ackerly 2001). Therefore it could be expected that invasive plants would have different flammability compared to native plants. This was evident in the study described in Chapter 2 where certain components of flammability of fuel from African Olive was different from fuel from native eucalypt woodland. Cootamundra Wattle is a native species from eucalypt woodland and it developed under the same broad evolutionary conditions as the woodlands it invaded. It was hypothesised that the flammability of Cootamundra Wattle would not differ greatly compared to fuel from YellowBox-Red Gum Grassy Woodland.

The presence of Cootamundra Wattle in the Red Hill Nature Reserve seems to be following the expected pathway found for other invasive woody weeds by changing the vertical distribution of fine fuel and consequently increasing the fire hazard. These changes could result in a positive feedback loop where the invasive species alter the environment favouring its own regeneration (Buckley *et al.* 2007). Cootamundra Wattle can recover after fire with mass germination of seed (Smith 1993; Morgan *et al.* 2002) and, together with short development time to reach maturity and high input of seed to the soil bank (Morgan *et al.* 2002), has the potential to become a highly invasive species. These results suggest that native weeds from similar habitats may not alter flammability but can change fuel loads and perhaps also fire behaviour.

4. Leaf functional traits and consequences for fire in eastern Australian forests and woodlands

4.1. Introduction

A plant functional trait is a term that has been broadly used to describe a plant attribute that can be measured for an individual but is relevant at an ecological organisation scale (Cunningham *et al.* 1999). Pérez-Harguindeguy *et al.* (2013) define plant functional traits as the characteristics (morphological, physiological, phenological) that constitute ecological strategies shaping plant responses to environmental factors, affecting other trophic levels and influencing ecosystem attributes. Drenovsky *et al.* (2012) state more simply that plant traits are measurable properties that can be scaled to populations, communities or ecosystems.

Plant functional traits have been shown to have strong connections with ecosystem processes while interactions and trade-offs among plant traits have been a long-standing focus of plant ecological studies (Diaz *et al.* 2004; Wright *et al.* 2004; Wright *et al.* 2007; Chapin *et al.* 2008; Donovan *et al.* 2011; Cardinale *et al.* 2012). There is an vast number of studies involving plant functional traits in a wide range of fields of science including botany, agriculture and forestry, and in different disciplines including conservation, evolution and ecology (Dickinson and Kirkpatrick 1985; Cunningham *et al.* 1999; Fernández-Escobar *et al.* 1999; Diaz *et al.* 2004; Wright *et al.* 2004; Dibble *et al.* 2007; Wright *et al.* 2007; Chapin *et al.* 2008; Pickett *et al.* 2009; Keeley *et al.* 2011; Schwilk and Caprio 2011; Cardinale *et al.* 2012; Dimitrakopoulos *et al.* 2013; Pérez-Harguindeguy *et al.* 2013).

An array of functional traits have been recognised as determinants of species tolerance such that there is a feedback cycle between some functional traits and environmental processes (Scarff and Westoby 2006). For example, fire influences plant community composition and structure and fire is influenced by vegetation structure and composition, resulting in a complex relationship (Mandle *et al.* 2011). The flammability of individual plant species varies greatly and, depending on the composition of the fuel bed, can determine the characteristics of a single fire by influencing fire intensity and flame height (Whelan 1995). The vegetation can therefore determine the fire regime for a given area (Gill and Moore 1996; Pausas *et al.* 2004; Gill and Zylstra 2005; Scarff and Westoby 2006; White and Zipperer 2010). Plant flammability relates to a set of traits that can be measured (Pérez-Harguindeguy *et al.* 2013) to give scientists an insight into fire ecology but can also provide land managers with knowledge on fuel hazard rating to improve fire management planning and fire behaviour prediction (Anderson 1970; Dimitrakopoulos and Papaioannou 2001).

The general concept of flammability is defined by how easily a material will ignite and burn. From a technical point of view, flammability is composed of four different components: ignitability, sustainability, combustibility and consumability (Anderson, 1970; Martin *et al*. 1994). At the individual plant level, 'ignitability' is the time elapsed until ignition on exposure to a heat source; 'sustainability' is the ability to sustain fire once ignited; 'combustibility' is the rate of burn after ignition and; 'consumability' is the proportion of mass or volume consumed by fire. Flammability can be related to fire characteristics at an ecosystem level (Anderson, 1970; Martin *et al*. 1994) such that the ignitability of individual plants drives the pattern of ignition in an ecosystem. Sustainability is related to the rate of fire spread and consumability is related to fire intensity. The consumability of vegetation is equivalent to the fuel load available for burning. These four components together with plant architecture will affect fire behaviour and fire intensity of planned or unplanned fires (Madrigal *et al.* 2009; Marino *et al.* 2010).

Leaves are arguably the most important flammable structure of a plant (Gill and Moore 1996; Etlinger and Beall 2005; Murray *et al.* 2013) as they are the first part of a plant to burn (Pickett *et al.* 2009; Murray *et al.* 2013). There are few studies relating plant functional traits such as leaf size and area with flammability (Schwilk and Caprio 2011; Murray *et al.* 2013), and at present there is no standardisation among methodologies to allow complex comparisons between species (Dimitrakopoulos 2001; Etlinger and Beall 2005; Scarff and Westoby 2006). Flammability has been shown to be associated with leaf length (Schwilk and Caprio 2011), width (Scarff and Westoby 2006), specific leaf area (SLA), thickness, moisture content and mass (Gill and Moore 1996; Dibble *et al.* 2007; Murray *et al.* 2013). Several studies have linked the mineral and heavy metal content of leaves to plant flammability (Philpot 1970; Gill and Moore 1996; Scarff and Westoby 2006). For example, the presence of large quantities of phosphorus (P) in leaves has been shown to influence leaf flammability by promoting high auto-ignition temperature (Scarff and Westoby 2008).

It is well accepted that fire regimes can be altered due to biological invasion (Mack and D'Antonio 1998; Williams and Baruch 2000; Brooks *et al.* 2004; Dibble *et al.* 2007; Pauchard *et al.* 2008; Rew and Johnson 2010; Allen *et al.* 2011; Berry *et al.* 2011). Although there is evidence suggesting that exotic species can increase fire intensity and fire spread (Berry *et al.* 2011; Murray *et al.* 2013), the impact of woody plants on fire regimes can vary significantly depending on the species involved (Mandle *et al.* 2011). Invasive plants change the nature of the fuel available for burning by presenting a different fuel arrangement (or architecture) compared to native vegetation but also by introducing different amounts and types of fuel with different chemical composition (Daehler 2003; Wright *et al.* 2007; Dawson *et al.* 2009; Fisher *et al.* 2009). The characterisation of leaf functional traits is therefore expected to be an effective method for determining the impact of invasive plant species on fire behaviour and fire regimes. Fernandes and Cruz (2012) argue that assessment of flammability under laboratory conditions is limited mostly by the scale of experimentation used and the difficulty in replicating the fuel bed found in natural conditions. Added to these limitations is the restriction that heat exposure is not comparable with natural conditions. The methodological limitations associated with flammability measurements are considered in this chapter and a new approach to assess and compare flammability among species is presented.

This study involves an analysis of a range of common leaf traits, the mineral composition of leaves and their relationship to the four components of flammability. In order to investigate these aspects invasive and native vascular plants species from woodland forests of eastern Australia were used. The hypothesis that invasive plants have different intrinsic fuel properties from those found in native species (DeBano *et al.* 1998; Brooks *et al.* 2004) was tested. To test this hypothesis the following research questions were formulated:

1. Do native and invasive plants differ in their leaf morphology and leaf flammability traits?
2. Are leaf morphology and leaf flammability traits related?

4.2 Material and methods

4.2.1 Site and species selection

Sites were selected opportunistically depending on the presence of woody weeds in areas of intact native bushland in the Sydney Basin that were accessible and not legally protected in some way. A detailed description of the geology, climate and vegetation of the Sydney Basin bioregion can be found in Interim Biogeographic Regionalisation for Australia (IBRA7 2014) but a brief description is provided here. The Sydney Basin bioregion covers an area of approximately 44 000 km^2 and represents about 4.5% of the area of New South Wales. It is located on the central east coast of New South Wales and extends from Batemans Bay northwards to Nelson Bay and westward to Mudgee. This bioregion includes a variety of landscapes mostly formed from sedimentary shale and sandstone and includes a range of topography and climates resulting in a variety of vegetation communities. Soil types vary from sandy soils in coastal areas to more developed coloured soils and well developed podsol types. The main native vegetation types sampled were the Cumberland Plains Woodland, Sydney Coastal Dry Sclerophyll Forest and Sydney Hinterland Dry Sclerophyll Forest (Keith 2004). Collection sites included Mount Annan (34° 3' 53.92" S, 150° 46' 13.40" E), Lawson (33° 43 '4.67" S, 150° 25' 38.76" E), Glenbrook (33° 46' 11.28" S, 150° 37' 14.21" E), Springwood (33° 41' 50.43" S, 150° 34' 6.62" E), Picton (34° 11' 12.67" S, 150° 36' 40.54" E) and Concord (33° 50' 32.22" S, 151° 5' 58.03" E). The climate for the Sydney Basin is mostly temperate with warm summers and no distinct dry season. A sub-humid climate occurs across large areas in the northeast, and a small area in the west of the bioregion around the Blue Mountains falls in a montane climate zone (Lawson, Glenbrook and Springwood). The mean annual temperature ranges from 10–17 °C with minimum average monthly temperatures ranging from -1.4–8.1 °C and maximum average monthly temperatures varying from 22.4–31.9 °C. The annual rainfall varies from 522–2395 mm (IBRA7 2014).

The woody weed species used in this study (Table 4.1) were considered to be invasive if they: (1) were generally targeted for treatment in fuel beds (e.g. Blackberry, Lantana, Privet); (2) are native to Australia but have become invasive outside of their native range (e.g. Cootamundra Wattle), or (3) are widely recognised as a non-native pest plant and are listed on the list of Weeds of National Significance (WoNS). The range of native plant species selected was based on their dominance in the sub-canopy of a given vegetation type (e.g. *Angophora costata*), ecological importance (e.g. *Banksia serrata*) and distribution (e.g. *Acacia rubida*) in terms of proportion of the fuel load within the forest stratum (Table 4.1).

Green, fully expanded leaves (at least 30 leaves per individual) were collected randomly from three mature plants of each species and stored in sealed plastic bags until processing. Additional leaf material was collected from the same three individuals but was bulked into a single sample to ensure that there was enough material for determination of flammability traits and mineral composition. The samples did not include branches, twigs, roots, cones, flowers or fruits.

4.2.2 Flammability traits

Flammability was assessed using a mass-loss calorimeter (MLC; Fire Testing Technology; UK). The MLC consists of a conical heater capable of producing radiative fluxes between 10 and 100 kW m^{-2} and a load cell to measure the change in mass of a sample over time. The cone heater and load cell are contained within a stainless steel enclosure, which is supplied with compressed air at a flow rate of 140 L min^{-1}. A 60 cm stainless steel chimney on top of the enclosure contains thermocouples that are calibrated using high purity (99%) methane gas (BOC Ltd, North Ryde, NSW, Australia) to quantify heat release as described in the standard ISO 13927 (ISO, 2001). In the MLC, a sample holder (10 ×10 × 5 cm) with a porosity of 27% was used to allow diffusion of air through the fuel samples.

Table 4.1. Invasive and native plant species used in this study and location of collection.

Species and authority	Family	Common name	Location
Invasive species			
Acacia baileyana F.Muell.	Fabaceae	Cootamundra Wattle	Gleenbrook
Acer negundo L.	Sapindaceae	Box Elder	Gleenbrook
Ageratina adenophora (Spreng.) King & H.Rob.	Asteraceae	Crofton Weed	Picton
Bambuseae spp.	Poaceae	Bamboo	Picton
Cestrum parqui L'Hér.	Solanaceae	Green Cestrum	Concord
Chrysanthemoides monilifera (L.) Norlindh	Asteraceae	Boneseed	Picton
Cinnamomum camphora (L.) J.Presl.	Lauraceae	Camphor Laurel	Picton
Cotoneaster coriaceus Franch.	Rosaceae	Cotoneaster	Lawson
Cystisus scoparius (L.)Link	Fabaceae	Scotch Broom	Picton
Erythrina crista-galli L.	Fabaceae	Brazilian Coral Tree	Picton
Hedera helix L.	Araliaceae	English Ivy	Picton
Lantana camara L.	Verbenaceae	Lantana	Gleenbrook
Ligustrum lucidum W.T.Aiton	Oleaceae	Broad-leaf Privet	Lawson
Ligustrum sinense Lour.	Oleaceae	Common Privet	Springwood
Ligustrum vulgare L.	Oleaceae	Chinese Privet	Gleenbrook
Lonicera japonica Thunb.	Caprifoliaceae	Japonese Honeysuckle	Lawson
Lycium ferocissimum Miers	Solanaceae	African Boxthorn	Picton
Olea europea subsp. *cuspidata* (Wall. Ec G.Don)	Oleaceae	African Olive	Mount Annan
Pinus radiata D. Don	Pinaceae	Radiata Pine	Camden
Pittosporum undulatum Vent.	Pittosporaceae	Pittosporum undulatum	Lawson
Prunus cerasifera Ehrh.	Rosaceae	Cherry Plum	Lawson
Pyracantha sp.	Rosaceae	Firethorn	Picton

Table 4.1. (cont.)

Species and Authority	Family	Common name	Location
Rhamnus alaternus L.	Rhamnaceae	Buckthorn	Lawson
Rhus typhina L.	Anacardiaceae	Staghorn Sumac	Glenbrook
Ricinus communis L.	Euphorbiaceae	Castor Oil Plant	Concord
Rubus fruticosus agg. L.	Rosaceae	Blackberry	Lawson
Senna pendula var. *glabrata* Willd. Vogel	Caesalpiniaceae	Easter Cassia	Glenbrook
Solanum mauritianum Scop.	Solanaceae	Wild Tobacco	Concord
Ulex europeus L.	Fabaceae	Gorse	Picton
Vinca major L.	Apocynaceae	Bigleaf Periwinkle	Lawson
Native species			
Acacia implexa Benth.	Fabaceae	Lightwood	Glenbrook
Acacia parramattensis Tindale	Fabaceae	Parramatta Wattle	Lawson
Acacia rubida (A.Cunn.) Pedley	Fabaceae	Red-stemmed Wattle	Lawson
Angophora costata (Gaertn.) Britten	Myrtaceae	Smooth-barked Apple	Lawson
Backhousia myrtifolia Hook. & Harv.	Myrtaceae	Grey Myrtle	Lawson
Banksia serrata L.f.	Proteaceae	Old Man Banksia	Lawson
Callicoma serratifolia Andrews	Cunoniaceae	Black Wattle	Lawson
Myrsine variabilis (R.Br.) Mez	Myrsinaceae	Muttonwood	Springwood
Smilax australis R.Br.	Smilacaceae	Lawyer Vine	Springwood

For each species, three samples were oven-dried at 100 °C until constant weight was reached. The weight of each sample varied according to the species and leaf shape. The samples were trimmed to fit the holder to uniformly cover the exposed surface area and sample thickness was maintained at 5 cm. Burns were done in triplicate using an irradiance of 25 kW m^{-2} and a 10 kV spark igniter was used to provide piloted ignition. Heat rate release (HRR; kW m^{-2}) and mass loss rate (MLR; gs^{-1}) were recorded at 1 Hz and the time-to-ignition and flameout was recorded manually. The average effective heat of flaming combustion (AEHC; MJ kg^{-1}) was calculated as the total heat release divided by the mass loss (MLCCalc; Fire Testing Technology, UK).

Outputs from the MLC were related to the components of flammability as defined by Anderson (1970) and Martin *et al*. (1994). Ignitability was determined by measuring the time-to-ignition; sustainability was assessed from the duration of flames; combustibility was considered to be equivalent to the mass-loss rate (burning rate) and; consumability was regarded as the residual mass fraction of the material burnt. Average effective heat of combustion is a measure of 'real world' heat of combustion (the energy produced by combusting a substance in air) and was used in conjunction with data from the literature to determine the effect of invasive species on estimates of fireline intensity.

To determine the gross heat of combustion of each species, a subsample of leaves (10 g) was oven-dried at 100 °C until constant weight was reached and finely ground using a bench grinder (MZ1000, RETSCH, Germany). One bulked sample per species was combusted in an oxygen bomb calorimeter (6400 Automatic Isoperibol Calorimeter, Parr Instrument Company, Illinois, USA). In a bomb calorimeter, electrical energy is used to ignite the fuel; as the fuel is burning, it will heat up the surrounding air, which expands and escapes through a tube that leads the air out of the calorimeter. When the air is escaping through the copper tube it will also heat up the water outside the tube. The temperature of the water allows for calculating calorie content of the fuel. The gross heat of combustion (HoC) can be used as an accurate measure to calculate the fire intensity through the equation presented by Byram (1959).

The instrument calculates the gross heat of combustion by:

$$Hc = \frac{WT - e1 - e2 - e3}{m} \qquad [4.1]$$

where:

Hc = gross heat of combustion (J g^{-1})

T = observed temperature rise (°C)

W = Energy equivalent of the calorimeter in calories per °C. The energy equivalent is determined by standardizing the calorimeter. In this case with 1 g of Benzoic Acid.

e1 = heat produced by burning the nitrogen portion of the air trapped in the bomb to form nitric acid (°C)

e2 = heat produced by the formation of sulphuric acid from the reaction of sulphur dioxide, water and oxygen (°C)

e3 = heat produced by heating wire and cotton thread (°C)

m = sample mass (g)

4.2.3. Leaf traits

Fully expanded leaves from three mature plants (n = 20) from each species were used to determine fresh weight and leaf dimensions. Length, width and thickness were measured with a digital caliper to the nearest 0.001 mm. Leaf surface area was measured using a LI-COR portable leaf area meter (LI-3000C, Lincoln, USA) fitted with a LI-3050C Transparent Belt Conveyor Accessory. The petiole was included in the measurement as recommended by Westoby (1998). The fresh weights of the same leaves was measured to the nearest 0.001 g (PB303-S Mettler Toledo Delta Range® balance, Mettler Toledo Ltd., Australia) prior to oven-drying at 60 ˚C to constant weight for calculation of moisture content. The specific leaf area (SLA) was calculated by dividing the leaf surface area per dry mass for each individual leaves.

4.2.4. Leaf chemistry

A subsample of leaves (n = 20) collected for each species was oven-dried at 60 ˚C to constant weight and ground to a fine powder in a mortar grinder (MZ1000, RETSCH, Germany) and analysed for total carbon (C) and nitrogen (N) by dry combustion (Elementar Vario Max CNS Analysensysteme GmbH, Hanau, Germany). The same leaf samples were used for determination of major and minor nutrients (Al, B, Ca, Cu, Fe, K, Mg, Mn, Na, P, S and Zn). The samples were analysed by a commercial company (CSBP Soil and Plant Analysis Lab, Bibra Lake, Western Australia) after digestion with 5 mL of nitric acid using a Milestone Ethos-1 microwave digester (Milestone Inc., USA). The elemental concentrations in the digests were determined by atomic absorption spectrometry (Vista Pro Inductively Coupled Plamsa-Optical Emission Spectrophotometer (ICP-OES), Varian Analytical Instruments, Palo Alto, USA).

4.2.5. Statistical analysis

A Pearson's correlation matrix analysis was used to explore and determine whether there were relationships among leaf traits [leaf surface area (LSA), thickness, length, width, leaf dry mass (LDM) and specific leaf area (SLA)], and flammability traits [gross heat of combustion (HoC), time to ignition (TTI), flame duration (FD), residual mass fraction (RMF), mean heat rate release (Mean HRR), peak heat rate release (peak HRR), mean energy heat of combustion (mean EHC) and peak mass loss rate (peak MLR)]. The relationships identified by the Pearson's correlation matrix were then plotted and a regression curves were fitted in order to explain the strength of relationships.

A factor analysis (Principal Component Analysis (PCA); rotation method Varimax with Kaiser normalisation (SPSS version 20) was done to explore the relationships among native and invasive species and flammability and leaf traits. This analysis attempted to identify underlying variables capable of explaining the distribution of the data points in two groups (native and invasive species). The factor analysis was used to reduce the data pool to a small number of factors capable of explaining most of the variance in the data. Any variable that did not meet the criteria of sampling adequacy (larger than 0.50) was removed from the analysis and the analysis was repeated. To visualise whether the groups had a consistent composition or showed variable patterns, a cluster analysis of the flammability-leaf traits factors and species relationship was done by constructing dendrograms using Ward's inertia method (SPSS version 20)

One-way ANOVA was done using the condition (native or invasive species) and factor scores from the PCA to confirm similarities and differences. The same analyses were made comparing the groups resulting from the dendograms and the PCA factor scores to identify any possible patterns between species and flammability and/or leaf traits.

To rank the flammability of each species tested, a multi-criteria analysis of flammability traits was done. Each component of flammability can be directly related to a measurement done by the mass loss calorimeter. Time to ignition relates to ignitability with the shortest time indicating greater flammability. Flame duration relates to sustainability with the longest time indicating greater residence time of fire. Similarly, mass loss rate relates to combustibility such that a faster rate indicates greater combustibility. Consumability is directly related to the residual mass fraction of the fuel remaining therefore smaller values indicate greater fuel consumption. On the basis of these rules, data was normalised to a linear scale of zero to 100. For example, the species with leaves that ignited the fastest was assigned a score of 100 and the species with leaves that took the longest time to ignite was assigned a score of 0 with the remaining species scaled between these two extremes. These scores were multiplied by a weighting factor and added together to produce an overall score which was then ranked. As it is uncertain what the weightings for flammability components should be, the same weight was given to each component (although it could be argued that without ignition the others are irrelevant). A Monte-Carlo analysis in which the input parameters values were substituted by probability density functions (PDFs) was done. The input values of the scaled TTI, FD, MLR and RMF parameters were fitted by using normally distributed PDFs constrained between the minimum and maximum values measured for each species. The weightings were fitted to a skewed Gaussian distribution, constrained between 0 and 1, that kept the mean weighting for each flammability parameter at 0.25. This was done to ensure that the average sum of weights over the simulations remained at 1. An *a priori* estimation of the initial number of iterations for the Monte-Carlo analysis needed to produce an analysis where the true mean of the distribution lies within 1% of the estimate was 45 000. After the analysis, the true error of the estimated mean was calculated as 0.38%. For each iteration of the Monte-Carlo analysis, the species were ranked in order of their flammability based on the weighted score calculated for each species. The frequency that each species occurred at each rank was calculated as a proportion of the total number of iterations in the simulation.

4.4. Results

4.4.1. Leaf functional traits and flammability

The size of leaves of invasive species ranged over two orders of magnitude with the smallest leaf being less that 2 cm^2 (*Acacia baileyana*) to over 500 cm^2 (*Ricinus communis*; Table 4.2). The leaves of native species were more moderately-sized with the largest leaf being just over 30 cm^2 (*Calicoma seratifolia*) and the smallest around 0.6 cm^2. Leaf thickness ranged from 0.12 to 0.48 mm and there were no significant differences between natives and invasive species (one-way ANOVA, P = 0.450). Leaf length ranged from 1.37 to 26.16 cm but there were no statistical differences between natives and invasive species (one-way ANOVA, P = 0.851).

Of the flammability-related traits measured (i.e. TTI, FD, peak MLR, mean EHC, mean HRR, peak HRR, RMF and HoC), only two were found to be statistically different between invasive and native species (Table 4.3). The gross heat of combustion of invasive species ranged from 16.5 to 21.1 MJ kg^{-1} while native species had a range from 17.1 to 22.0 MJ kg^{-1} (one-way ANOVA, P = 0.020). The residual mass fraction was also statistically different between native and invasive species (one-way ANOVA, P = 0.030) and from 3–21% of the original mass was left after combustion of invasive plant material while the residual mass fraction for native species ranged from 6–19%.

Leaf chemistry analysis showed large variation in concentration of N, B, Ca, Cu, K, Mg, P, S, and Zn among species (Table 4.4). The proportion of P in leaves of invasive species ranged from 0.07–0.43% and from 0.02–0.12% for native species. Leaves of invasive species also had higher proportions of K ranging from 0.62–4.20% while for natives species the range was much smaller (0.16–0.90%). When grouped together, the invasive species had significantly higher proportions of N (one-way ANOVA, $P = 0.008$), B ($P = 0.002$), Ca ($P = 0.002$), Cu ($P = 0.013$), K ($P = 0.005$), Mg ($P = 0.006$), P ($P = 0.001$), S ($P = 0.018$) and Zn ($P = 0.051$) compared to native species.

Table 4.2. Plant functional traits including leaf surface area, thickness, length, width, leaf dry mass and specific leaf area from invasive and native species occurring in the study sites in New South Wales, Australia. Values are mean ± standard deviation (n = 20).

	Leaf surface area (cm^2)	Thickness (mm)	Length (cm)	Width (cm)	Leaf dry mass (g)	Specific leaf area (m^2 kg^{-1})
Invasive species						
Acacia baileyana	1.90 ± 0.29	0.21 ± 0.04	2.43 ± 0.24	1.26 ± 0.22	0.018 ± 0.054	14.87 ± 4.32
Acer negundo	26.62 ± 8.76	0.27 ± 0.06	9.95 ± 1.52	4.68 ± 1.12	0.289 ± 0.011	9.59 ± 1.56
Ageratina adenophora	35.75 ± 6.07	0.27 ± 0.05	10.72 ± 2.32	6.61 ± 0.73	0.115 ± 0.149	6.20 ± 1.19
Bambuseae spp.	16.45 ± 6.02	0.12 ± 0.02	11.97 ± 3.16	1.98 ± 0.28	0.095 ± 0.030	20.76 ± 2.72
Cestrum parqui	28.86 ± 7.22	0.17 ± 0.02	14.12 ± 1.84	3.68 ± 0.52	0.075 ± 0.097	7.55 ± 0.75
Chrysanthemoides monilifera	6.59 ± 3.47	0.32 ± 0.04	5.93 ± 0.77	1.95 ± 0.61	0.061 ± 0.093	3.68 ± 0.37
Cinnamomum camphora	26.27 ± 9.90	0.24 ± 0.03	10.16 ± 2.45	4.52 ± 0.95	0.286 ± 0.141	7.82 ± 0.42
Cotoneaster coriaceus	8.99 ± 2.28	0.27 ± 0.03	5.42 ± 0.66	2.62 ± 0.37	0.115 ± 0.032	7.47 ± 0.99
Cytisus scoparius	0.58 ± 0.13	0.17 ± 0.01	1.37 ± 0.27	0.85 ± 0.48	0.003 ± 0.002	8.90 ± 1.75
Erythrina crista-galli	91.88 ± 41.60	0.22 ± 0.04	13.36 ± 4.68	11.44 ± 3.52	0.267 ± 0.313	9.72 ± 1.88
Hedera helix	49.61 ± 26.35	0.33 ± 0.06	8.98 ± 2.68	8.30 ± 2.45	0.501 ± 0.397	6.56 ± 0.70
Lantana camara	32.34 ± 14.54	0.27 ± 0.06	8.93 ± 2.13	5.38 ± 1.38	0.126 ± 0.182	8.89 ± 1.11
Ligustrum lucindum	27.89 ± 9.14	0.21 ± 0.02	13.31 ± 2.50	3.44 ± 0.59	0.255 ± 0.150	6.96 ± 0.76
Ligustrum sinense	28.20 ± 8.52	0.29 ± 0.04	9.67 ± 1.88	4.49 ± 0.83	0.262 ± 0.201	6.01 ± 1.76
Ligustrum vulgare	3.70 ± 1.93	0.15 ± 0.03	3.34 ± 1.01	1.64 ± 0.36	0.017 ± 0.025	10.24 ± 1.69
Lonicera japonica	14.77 ± 4.56	0.23 ± 0.04	7.59 ± 1.32	3.15 ± 0.53	0.077 ± 0.095	8.30 ± 3.19
Lycium ferocissimum	1.83 ± 0.49	0.35 ± 0.07	2.65 ± 0.48	1.32 ± 0.39	0.009 ± 0.031	2.89 ± 0.45
Olea europea sp. *cuspidata*	8.43 ± 2.90	0.33 ± 0.05	8.22 ± 1.26	1.69 ± 0.39	0.123 ± 0.041	4.87 ± 0.98

Table 4.2. (cont.)

	Leaf surface area (cm^2)	Thickness (mm)	Length (cm)	Width (cm)	Leaf dry mass (g)	Specific leaf area (m^2 kg^{-1})
Pittosporum undulatum	14.54 ± 2.79	0.24 ± 0.03	9.43 ± 0.97	2.61 ± 0.31	0.189 ± 0.050	6.91 ± 0.54
Prunus cerasifera	18.10 ± 4.73	0.25 ± 0.04	7.89 ± 1.11	3.83 ± 0.61	0.132 ± 0.729	10.65 ± 2.67
Pyracantha sp.	1.95 ± 0.61	0.24 ± 0.03	3.19 ± 1.02	1.25 ± 0.47	0.024 ± 0.009	8.11 ± 1.14
Rhamnus alaternus	5.48 ± 2.97	0.33 ± 0.05	4.34 ± 1.11	1.91 ± 0.51	0.065 ± 0.074	5.41 ± 1.20
Rhus typhina	31.19 ± 7.22	0.16 ± 0.03	12.24 ± 1.95	4.00 ± 0.57	0.120 ± 0.081	10.58 ± 1.09
Ricinus communis	553.21 ± 128.11	0.22 ± 0.03	26.16 ± 3.17	34.78 ± 3.97	4.044 ± 2.643	6.12 ± 0.57
Rubus fruticosus agg.	12.13 ± 4.28	0.22 ± 0.05	5.57 ± 1.61	3.20 ± 0.59	0.073 ± 0.049	9.37 ± 1.01
Senna pendula var. *glabrata*	6.51 ± 2.10	0.15 ± 0.04	4.69 ± 1.27	1.95 ± 0.18	0.025 ± 0.020	11.85 ± 1.46
Solanum mauritianum	147.64 ± 34.88	0.48 ± 0.04	24.42 ± 3.09	9.68 ± 1.08	0.916 ± 0.545	6.66 ± 0.59
Vinca major	11.09 ± 3.01	0.33 ± 0.06	6.28 ± 0.88	3.04 ± 0.50	0.120 ± 0.080	4.26 ± 0.39
Native species						
Acacia implexa	12.26 ± 4.27	0.27 ± 0.03	12.85 ± 1.9	1.70 ± 0.42	15.48 ± 0.096	5.97 ± 1.53
Acacia paramatensis	1.87 ± 0.44	0.19 ± 0.05	3.8 ± 0.51	0.91 ± 0.44	0.024 ± 0.005	7.92 ± 1.21
Acacia rubida	14.11 ± 4.09	0.36 ± 0.03	12.2 ± 1.52	1.94 ± 0.37	0.260 ± 0.085	5.43 ± 0.33
Angophora costata	22.94 ± 5.90	0.37 ± 0.04	13.42 ± 1.40	2.98 ± 0.65	0.352 ± 0.149	6.91 ± 1.21
Backhousia myrtifolia	13.74 ± 4.34	0.27 ± 0.03	7.67 ± 1.47	2.94 ± 0.50	0.182 ± 0.075	8.21 ± 3.34
Banksia serrate	15.87 ± 6.95	0.44 ± 0.06	9.08 ± 2.73	2.53 ± 0.54	0.239 ± 0.106	6.70 ± 0.72
Calicoma seratifolia	31.72 ± 16.33	0.35 ± 0.06	10.97 ± 2.22	4.38 ± 1.36	0.414 ± 0.221	7.81 ± 0.60
Myrsina variabilis	16.43 ± 6.21	0.33 ± 0.04	7.99 ± 1.62	3.17 ± 0.61	0.253 ± 0.160	6.98 ± 1.60
Smilax australis	13.07 ± 4.93	0.37 ± 0.05	6.23 ± 1.00	3.10 ± 0.67	0.191 ± 0.076	6.97 ± 0.91
Melaleuca sp.	0.61 ± 0.11	0.17 ± 0.01	2.15 ± 0.32	0.63 ± 0.25	0.006 ± 0.003	13.25 ± 14.71

Table 4.3. Time to ignition (TTI), flame duration (FD), peak mass loss rate (Peak MLR), mean energy heat of combustion (Mean EHC), mean heat rate release (Mean HRR), peak heat rate release (Peak HRR), residual mass fraction (RMF) and gross heat of combustion (HoC) of invasive and native species occurring in dry eucalypt forests and woodlands in NSW, Australia. Values are mean ± standard deviation (n = 3); Values for gross heat of combustion were based on a single bulked sample.

	TTI (s)	FD (s)	Peak MLR ($g\ s^{-1}$)	Mean EHC ($MJ\ kg^{-1}$)	Mean HRR ($kW\ m^{-2}$)	Peak HRR ($kW\ m^{-2}$)	RMF (%)	HoC ($MJ\ kg^{-1}$)
Invasive species								
Acacia baileyana	18 ± 8	278 ± 33	0.22 ± 0.17	17.99 ± 0.87	97 ± 7	155 ± 11	17 ± 3	21.14
Acer negundo	44 ± 16	39 ± 8	0.13 ± 0.01	13.83 ± 0.86	122 ± 17	149 ± 20	13 ± 1	18.20
Ageratina adenophora	14 ± 5	71 ± 19	0.20 ± 0.06	13.46 ± 0.32	133 ± 17	163 ± 14	15 ± 3	16.96
Bambuseae spp.	9 ± 1	52 ± 6	0.25 ± 0.03	13.53 ± 0.77	174 ± 10	214 ± 3	10 ± 2	18.38
Cestrum parqui	7 ± 1	54 ± 4	0.11 ± 0.02	15.40 ± 2.93	116 ± 3	139 ± 2	16 ± 2	17.12
Chrysanthemoides monilifera	12 ± 5	107 ± 3	0.19 ± 0.05	14.43 ± 0.87	149 ± 15	187 ± 19	11 ± 2	19.92
Cinnamomum camphora	6 ± 1	63 ± 9	0.13 ± <0.01	14.59 ± 0.35	139 ± 13	175 ± 10	9 ± 2	19.33
Cotoneaster coriaceus	9 ± 6	99 ± 20	0.17 ± 0.04	16.29 ± 3.36	153 ± 7	202 ± 5	12 ± 2	18.77
Cystisus scoparius	23 ± 2	417 ± 50	0.10 ± <0.01	18.42 ± 2.76	107 ± 4	151 ± 2	18 ± 2	19.53
Erythrina crista-galli	7 ± 6	70 ± 11	0.12 ± 0.02	16.43 ± 2.49	102 ± 4	126 ± 10	11 ± 2	18.51
Hedera helix	16 ± 13	98 ± 6	0.23 ± 0.12	15.90 ± 0.34	151 ± 11	194 ± 18	13 ± 2	18.05
Lantana camara	15 ± 1	17 ± 1	0.18 ± 0.07	9.56 ± 0.31	95 ± 2	158 ± 5	15 ± 3	18.31
Ligustrum lucindum	17 ± 1	66 ± 4	0.16 ± 0.02	16.35 ± 3.67	152 ± <1	188 ± <1	10 ± 1	18.80
Ligustrum sinense	8 ± 2	107 ± 23	0.14 ± 0.02	13.65 ± 1.10	153 ± 11	208 ± 12	12 ± 2	19.53
Ligustrum vulgare	11 ± 6	165 ± 28	0.13 ± 0.03	17.69 ± 0.98	123 ± 20	163 ± 17	14 ± 1	18.89
Lonicera japonica	12 ± 3	89 ± 11	0.13 ± <0.01	17.34 ± 0.41	131 ± 8	160 ± 10	13 ± 2	20.43
Lycium ferocissimum	58 ±6	211 ± 60	0.13 ± 0.05	17.59 ± 0.63	119 ± 26	147 ± 26	21 ± 3	20.98
Olea europea subsp. *cuspidata*	22 ± 3	163 ± 10	0.22 ± 0.03	18.59 ± 4.29	172 ± 11	261 ± 28	18 ± 8	20.21

Table 4.3. (cont.)

	TTI (s)	FD (s)	Peak MLR (g s^{-1})	Mean EHC (MJ kg^{-1})	Mean HRR (kW m^{-2})	Peak HRR (kW m^{-2})	RMF (%)	HoC (MJ kg^{-1})
Pinus radiata	33 ± 3	178 ± 16	0.17 ± 0.01	15.17 ± 1.49	138 ± 10	201 ± 12	15 ± 4	19.87
Pittosporum undulatum	11 ± 4	103 ± 15	0.13 ± 0.01	15.31 ± 1.73	140 ± 11	183 ± 15	11 ± 1	19.53
Prunus cerasifera	15 ± 3	68 ± 20	0.12 ± 0.02	17.01 ± 0.59	115 ± 12	142 ± 11	10 ± 3	18.60
Pyracantha sp.	15 ± <1	220 ± 28	0.21 ± 0.13	20.17 ± 1.72	152 ± 6	197 ± 12	14 ± 1	22.11
Rhamnus alaternus	24 ± 13	78 ± 7	0.15 ± 0.02	17.13 ± 0.96	151 ± 10	186 ± 14	14 ± 2	18.95
Rhus typhina	7 ± 6	82 ± 16	0.10 ± 0.02	12.86 ± 0.51	94 ± 8	116 ± 7	12 ± 2	18.38
Ricinus communis	21 ± 7	150 ± 22	0.10 ± 0.01	15.99 ± 0.80	98 ± 12	125 ± 15	21 ± 1	17.73
Rubus fruticosus agg.	24 ± 15	39 ± 13	0.25 ± 0.11	14.06 ± 0.91	146 ± 8	183 ± 14	8 ± 2	18.98
Senna pedula var. *glabrata*	18 ± 2	108 ± 11	0.24 ± 0.23	18.80 ± 2.62	124 ± 3	156 ± 6	16 ± 1	16.53
Solanum mauritianum	10 ± 2	120 ± 30	0.10 ± 0.01	17.78 ± 2.10	113 ± 26	145 ± 22	17 ± 4	18.32
Ulex europaeus	56 ± 11	20 ± 5	0.26 ± 0.04	12.23 ± 0.05	168 ± 26	221 ± 26	3 ± 1	18.45
Vinca major	19 ± 4	118 ± 9	0.14 ± <0.01	17.57 ± 1.96	167 ± 10	213 ± 17	11 ± 3	19.74
Native species								
Acacia implexa	17 ± 4	93 ± 23	0.15 ± 0.03	16.43 ± 0.41	150 ± 13	193 ± 15	8 ± 1	20.92
Acacia paramatensis	33 ± 9	372 ± 68	0.15 ± 0.03	16.22 ± 2.03	92 ± 8	146 ± 12	19 ± 2	20.61
Acacia rubida	24 ± 12	64 ± 7	0.17 ± 0.02	14.40 ± 2.26	140 ± 5	176 ± 9	7 ± 1	22.02
Angophora costata	19 ± 8	59 ± 1	0.34 ± 0.18	11.11 ± 3.05	157 ± 1	201 ± 9	7 ± 1	20.51
Backhousia myrtifolia	7 ± 4	57 ± 3	0.23 ± 0.05	14.88 ± 1.19	219 ± 33	283 ± 43	6 ± 1	19.89
Calicoma seratifolia	22 ± 16	64 ± 11	0.17 ± 0.02	14.93 ± 1.41	149 ± 11	187 ± 17	11 ± 2	19.13
Myrsina variabilis	8 ± 8	91 ± 11	0.17 ± 0.05	16.56 ± 0.74	157 ± 4	205 ± 9	6 ± 1	21.67
Smilax australis	108 ± 55	45 ± 7	0.17 ± 0.04	13.52 ± 2.52	151 ± 11	187 ± 13	10 ± 2	17.10

Table 4.4. Elemental composition of leaves of invasive and native species. (C = carbon, N = nitrogen, Ca = calcium, K = potassium, Mg = magnesium, P = phosphorous, S = sulphur, B = boron, Cu = copper, Fe = iron, Mn = manganese). Values are based on a single bulked sample of 20 leaves of each species.

	C (%)	N (%)	C:N ratio	Ca (%)	K (%)	Mg (%)	P (%)	S (%)	B (mg kg^{-1})	Cu (mg kg^{-1})	Fe (mg kg^{-1})	Mn (mg kg^{-1})
Invasive species												
Acacia baileyana	53.52	2.78	19.28	0.39	0.83	0.23	0.11	0.13	148.77	5.27	139.63	37.62
Acer negundo	47.79	1.23	38.76	2.41	0.93	0.31	0.09	0.18	209.98	6.24	131.63	70.12
Ageratina adenophora	43.79	2.43	18.03	1.20	2.01	0.79	0.16	0.21	153.62	10.7	2439.88	286.77
Bambuseae spp.	45.78	2.49	18.38	0.37	0.93	0.20	0.15	0.16	109.48	7.54	1018.26	80.43
Cestrum parqui	43.56	4.05	10.76	1.62	4.20	0.38	0.32	0.39	110.08	15.18	95.10	19.39
Chrysanthemoides monilifera	49.38	1.04	47.55	1.06	0.93	0.42	0.09	0.14	132.56	8.75	90.44	480.13
Cinnamomum camphora	50.02	2.38	21.01	1.10	1.23	0.34	0.13	0.18	70.26	8.17	116.31	237.72
Cotoneaster coriaceus	49.38	1.27	38.82	1.49	0.84	0.46	0.10	0.07	69.59	5.96	207.75	96.77
Cystisus scoparius	50.40	3.48	14.48	0.63	0.76	0.37	0.16	0.15	63.07	5.19	209.88	234.87
Erythrina crista-galli	46.62	2.99	15.61	1.60	0.76	0.82	0.15	0.20	89.06	4.98	139.08	90.96
Hedera helix	46.05	1.83	25.19	2.50	1.00	0.76	0.17	0.19	72.02	5.11	199.36	186.02
Lantana camara	46.82	2.52	18.59	1.75	1.95	0.53	0.15	0.19	84.75	17.58	239.50	157.37
Ligustrum lucindum	48.83	1.56	31.31	2.26	0.98	0.57	0.12	0.33	67.81	16.92	120.20	130.66
Ligustrum sinense	50.29	1.69	29.72	1.09	0.62	0.27	0.10	0.11	56.72	8.90	506.68	319.44
Ligustrum vulgare	47.36	2.11	22.48	1.25	1.07	0.50	0.12	0.20	54.66	8.27	663.58	346.80
Lonicera japonica	47.86	1.75	27.28	1.00	1.69	0.44	0.11	0.17	65.62	5.86	80.13	60.64
Lycium frocissimum	42.85	3.23	13.25	1.16	1.27	0.94	0.29	0.32	54.83	18.13	271.90	302.76
Olea europea subsp. *cuspidata*	50.00	1.66	30.12	0.94	0.71	0.12	0.15	0.11	42.72	7.59	130.75	22.43
Pittosporum undulatum	49.91	0.97	51.3	1.10	1.87	0.21	0.07	0.08	68.56	4.50	352.40	619.94

Table 4.4 (cont.)

	C (%)	N (%)	C:N Ratio	Ca (%)	K (%)	Mg (%)	P (%)	S (%)	B (mg kg^{-1})	Cu (mg kg^{-1})	Fe (mg kg^{-1})	Mn (mg kg^{-1})
Prunus cerasifera	49.24	1.23	39.99	1.47	0.97	0.28	0.07	0.1	48.83	4.82	178.60	102.64
Pyracantha sp.	50.42	1.78	28.39	1.03	0.65	0.28	0.12	0.11	45.44	6.11	214.07	46.01
Rhamnus alaternus	50.49	1.08	46.7	0.75	1.08	0.20	0.09	0.08	60.26	4.57	67.01	322.60
Rhus typhina	47.18	3.13	15.07	1.84	1.65	0.21	0.33	0.16	64.71	10.21	140.09	28.57
Ricinus communis	45.63	4.93	9.25	2.30	1.55	0.36	0.43	0.39	51.20	8.74	212.51	41.50
Rubus fruticosus agg.	48.60	2.28	21.32	1.07	1.05	0.48	0.15	0.14	55.75	7.20	106.73	229.31
Senna pedula var. *glabrata*	43.48	2.26	19.26	2.86	1.70	0.38	0.17	0.36	53.71	8.11	255.46	50.66
Solanum mauritianum	46.53	3.49	13.35	2.58	0.97	0.20	0.22	0.29	56.00	7.57	174.97	18.79
Ulex europeus	49.37	1.69	29.28	0.13	0.76	0.12	0.07	0.08	26.41	3.78	101.79	33.94
Vinca major	49.72	1.72	28.94	1.40	1.73	0.25	0.07	0.13	50.05	7.83	102.64	66.50
Native species												
Acacia implexa	52.41	2.37	22.12	0.49	0.81	0.39	0.09	0.13	29.33	10.5	294.52	234.29
Acacia paramatensis	52.86	2.76	19.19	0.65	0.9	0.25	0.12	0.15	31.83	5.65	118.69	58.78
Acacia rubida	51.72	1.16	44.59	0.36	0.57	0.21	0.07	0.10	27.36	2.95	47.30	246.36
Angophora costata	53.83	0.83	65.14	0.32	0.52	0.24	0.04	0.06	20.54	4.53	44.58	82.97
Backhousia myrtifolia	51.56	0.84	61.13	0.81	0.85	0.21	0.06	0.05	27.59	5.31	821.8	62.53
Banksia serrate	52.62	0.44	119.7	0.24	0.16	0.14	0.02	0.04	24.25	1.60	66.06	280.94
Callicoma seratifolia	50.18	1.12	44.92	0.98	0.44	0.18	0.05	0.22	23.91	4.95	85.02	443.81
Casuarina sp.	48.78	1.4	34.79	0.95	0.67	0.21	0.07	0.09	116.58	3.01	153.76	358.98
Melaleuca sp.	50.08	1.78	28.12	1.11	0.54	0.16	0.07	0.09	41.25	9.49	87.12	16.94
Myrsine variabilis	54.84	0.91	60.38	0.22	0.48	0.16	0.03	0.15	21.06	3.20	107.65	219.28
Smilax australis	58.84	1.23	47.9	0.97	0.86	0.10	0.04	0.10	4.03	3.97	40.69	137.15

4.4.2. Leaf morphology and flammability traits correlations

There were few strong relationships among leaf morphology and flammability traits (Table 4.5). Leaf length and flame duration were negatively correlated (Pearson's correlation; $P < 0.001$) for both native ($R^2 = 0.3488$) and invasive species ($R^2 = 0.5275$). When natives and invasive species were grouped together, the regression was also significant (Figure 4.1; $R^2 = 0.4927$). Mean EHC and leaf length were negatively correlated but the association for native species (Figure 4.2) was weak ($R^2 = 0.1219$), as was the association for invasive species ($R^2 = 0.3538$) and when all plants were grouped together ($R^2 = 0.3398$). Pearson's correlation identified a correlation between HoC and leaf width (Pearson's correlation; $P = 0.029$) However, the regression curves had low R^2 values for were considered too low for native species ($R^2 = 0.1466$), invasive species ($R^2 = 0.0817$) and when both groups were considered together ($R^2 = 0.1027$). A weak negative correlation (Pearson's correlation; $P = 0.017$) was also found between flame duration and leaf width. For this pairing, the regression curves (Figure 4.3) for native species ($R^2 = 0.6159$), invasive species ($R^2 = 0.462$) and both plant groups together ($R^2 = 0.4415$) were reasonably strong.

Table 4.5. Pearson correlation matrix for leaf and flammability traits of native and invasive species (HoC = Gross heat of combustion, RMF = Residual mass fraction, TTI = Time to ignition, Mean HRR = mean heat rate release, Peak HRR = peak heat rate release, Peak MLR = mass loss rate, Mean EHC = mean energy heat of combustion, LSA = leaf surface area, LDM = Leaf dry mass and SLA = specific leaf area). n = 36. Numbers in bold indicate significant interactions. * Correlation is significant at the 0.05 level (two-tailed). ** Correlation is significant at the 0.01 level (two-tailed).

		TTI (s)	FD (s)	Peak MLR ($g\ s^{-1}$)	Mean EHC ($MJ\ kg^{-1}$)	Mean HRR ($kW\ m^{-2}$)	Peak HRR ($kW\ m^{-2}$)	RMF (%)	HoC ($MJ\ kg^{-1}$)
LSA	Pearson correlation	-0.205	-0.262	-0.250	-0.122	-0.224	-0.283	0.037	-0.259
	Significance	0.231	0.123	0.141	0.477	0.189	0.095	0.829	0.127
Thickness	Pearson correlation	0.293	-0.248	0.074	-0.135	0.297	0.303	-0.237	0.100
	Significance	0.083	0.144	0.666	0.434	0.078	0.073	0.164	0.561
Length	Pearson correlation	-0.266	**-0.513****	-0.121	**-0.377***	0.006	-0.090	-0.264	-0.125
	Significance	0.117	0.001	0.484	0.023	0.972	0.601	0.120	0.466
Width	Pearson correlation	-0.193	**-0.395***	-0.196	-0.242	-0.198	-0.275	-0.032	**-0.364***
	Significance	0.261	0.017	0.251	0.154	0.248	0.104	0.852	0.029
LDM	Pearson correlation	-0.033	-0.056	-0.052	0.028	0.084	0.073	-0.208	0.200
	Significance	0.850	0.747	0.763	0.871	0.625	0.672	0.223	0.242
SLA	Pearson correlation	-0.190	0.019	0.210	-0.085	-0.181	-0.173	-0.007	-0.017
	Significance	0.268	0.914	0.220	0.622	0.290	0.314	0.966	0.924

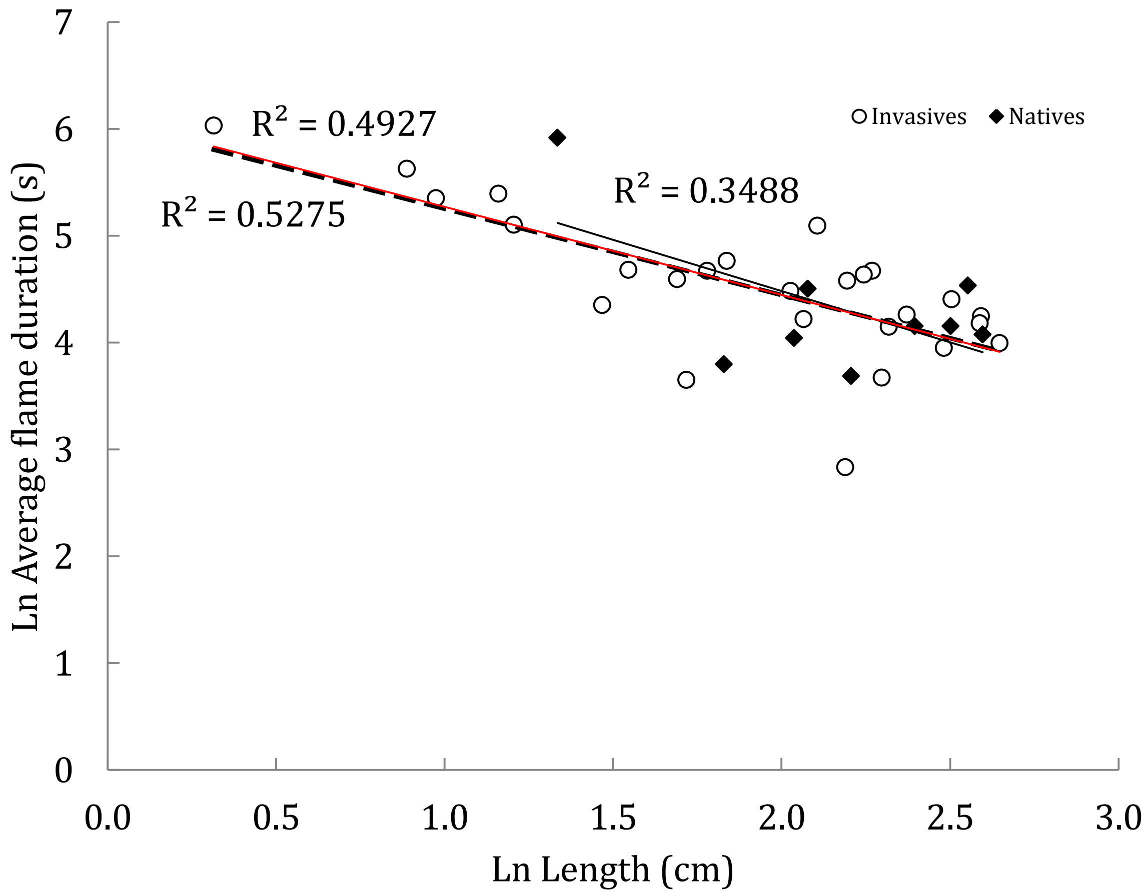

Figure 4. 1. Linear correlation between average flame duration (s; data L_n transformed) and leaf length (cm; data L_n transformed) of native and invasive species. (Dashed line = regression curve for invasive species; solid black line = regression curve for native species; solid red line = regression curve for both plant groups).

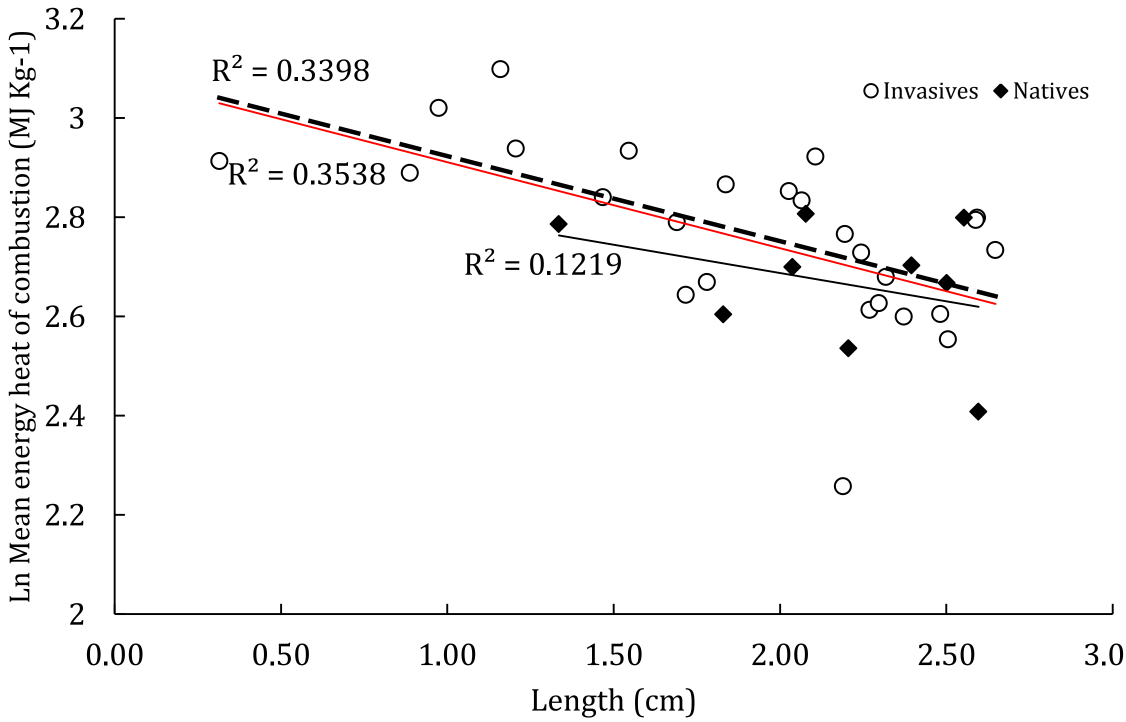

Figure 4. 2. Linear correlation between mean heat of combustion (kW m-^{2}; data L_n transformed) and leaf length (cm; data L_n transformed) of native and invasive species. (Dashed line = regression curve for invasive species; solid black line = regression curve for native species; solid red line = regression curve for both plant groups).

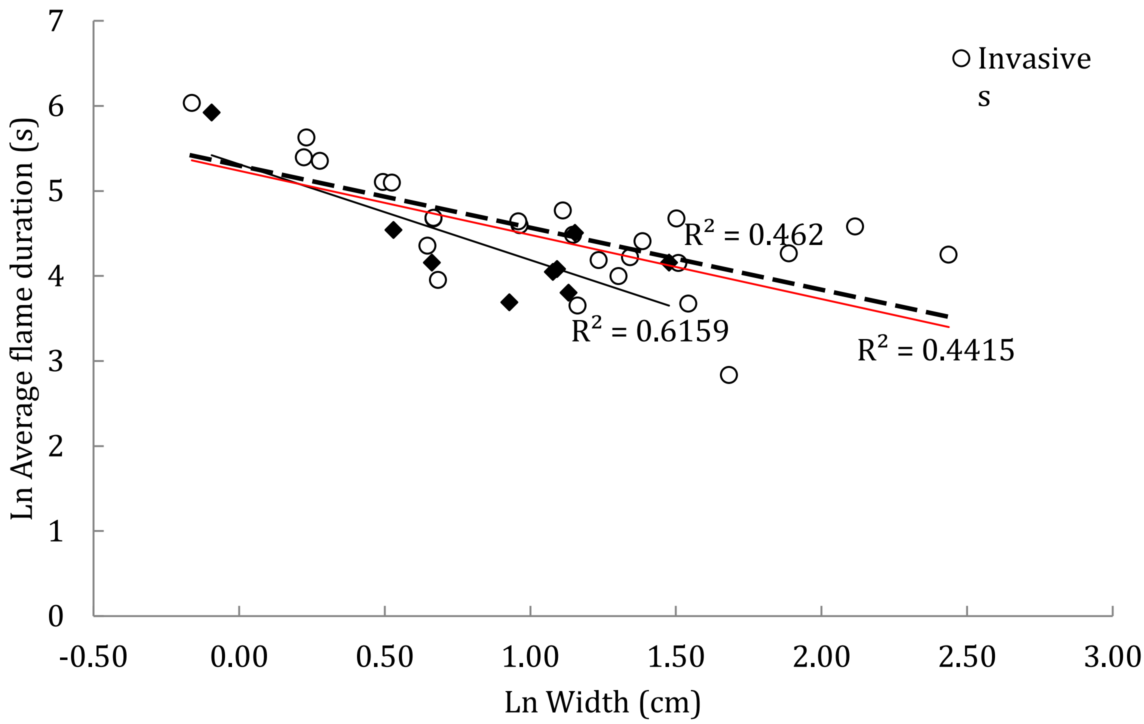

Figure 4. 3. Linear correlation between average flame duration (s; data L_n transformed) and leaf width (cm; data L_n transformed) of native and invasive species. (Dashed lines = regression curve for invasive species; solid black line = regression curve for native species; solid red line = regression curve for both plant groups).

4.4.3. Principal component analysis

The Principal Component Analysis (PCA) confirmed the separation of the species in two distinct groups: native and invasive species (Figure 4.4). The Kaiser-Meyer-Olkin measure of sampling adequacy (0.597) and Bartlett's test of sphericity (<0.001) indicated that the data set was appropriate for a factor analysis. The eigenvalues for Axes 1 and 2 were 3.807 and 2.556, respectively. PCA Axes 1 and 2 accounted for 35.63% and 34.89% of the total variance respectively. Further axes are not discussed as none accounted for more than 10% of the total variance.

The eigenvector values for Axis 1 ranged from -0.686 for RMF to 0.936 for mean HRR (Table 4.6). At the low score end of PCA Axis 1 were species with a higher RMF. At the high score end of Axis 1, species showed higher mean HRR, higher peak HRR and higher peak MLR. Invasive species tended to be located at the high end of PCA Axis 1 while natives were found distributed on the other extreme (Figure 4.5).

Axis 2 of the PCA was related to morphological attributes of leaves. Eigenvector values ranged from 0.494 for leaf thickness to 0.947 for LSA. Species with long and wide leaves and large surface area tended to be located at the higher end of Axis 2 of the PCA while species with small thin leaves were found on the opposite end (Figure 4.5).

Table 4.6. Rotated component matrix containing eigenvector scores for traits on the first two PCA axes. Eigenvector values >0.400 are in bold. (LSA = leaf surface area, RMF = residual mass fraction, HRR = heat release rate, MLR = mass loss rate).

Trait	PCA Axis 1	PCA Axis 2
LSA	-0.195	**0.947**
RMF	**-0.686**	-0.202
Mean HRR	**0.936**	-0.065
Peak MLR	**0.632**	-0.206
Peak HRR	**0.916**	-0.155
Thickness	0.436	**0.494**
Length	0.067	**0.879**
Width	-0.181	**0.876**

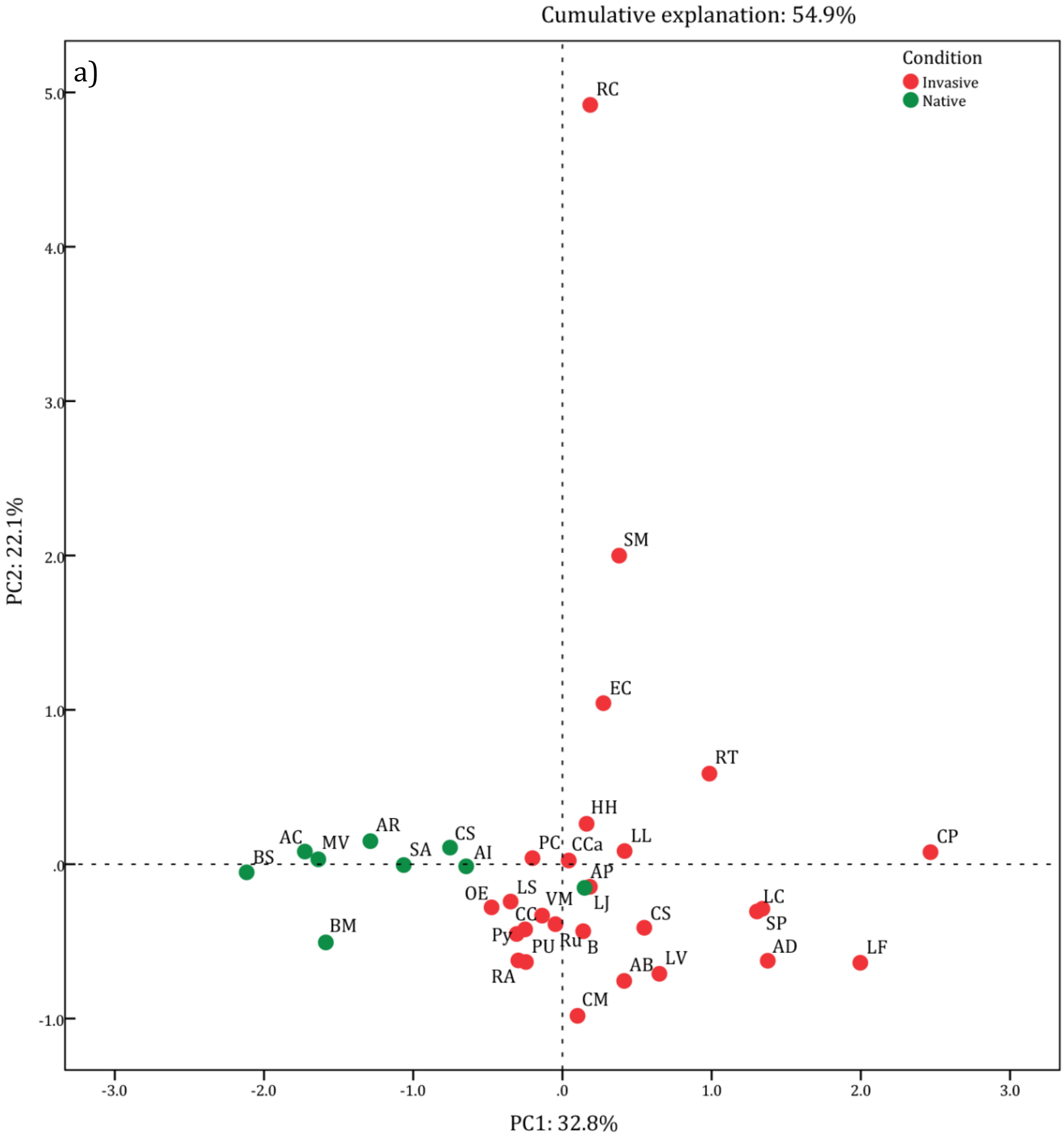

Figure 4. 4. Principal component analysis. LSA = leaf surface area, RMF = residual mass fraction, HRR = heat release rate, MLR = mass loss rate. AB, *Acacia baileyana*; AD, *Ageratina adenophora*; B, Bambuseae spp.; CP, *Cestrum parqui*; CM, *Chrysanthemoides monilifera*; CCa, *Cinnamomum camphora*; CC, *Cotoneaster coriaceus*; CS, *Cystisus scoparius*; EC, *Erythrina crista-galli*; HH, *Hedera helix*; LC, *Lantana camara*; LL, *Ligustrum lucindum*; LS, *Ligustrum sinense*; LV, *Ligustrum vulgare*; LJ, *Lonicera japonica*; LF, *Lycium ferocissimum*; OE, *Olea europea* sp. *cuspidata*; PU, *Pittosporum undulatum*; PC, *Prunus cerasifera*; Py, *Pyracantha* sp.; RA, *Rhamnus alaternus*; RT, *Rhus typhina*; Ru, *Rubus fruticosus* agg.; SA, *Smilax australis*; SP, *Senna pedula* var. *glabrata*; SM, *Solanum mauritianum*; VM, *Vinca major*; AI, *Acacia implexa*; AP, *Acacia paramatensis*; AR, *Acacia rubida*; AC, *Angophora costata*; BM, *Backhousia myrtifolia*; BS, *Banksia serrata*; CS, *Calicoma seratifolia*. • Invasive species, • Native species.

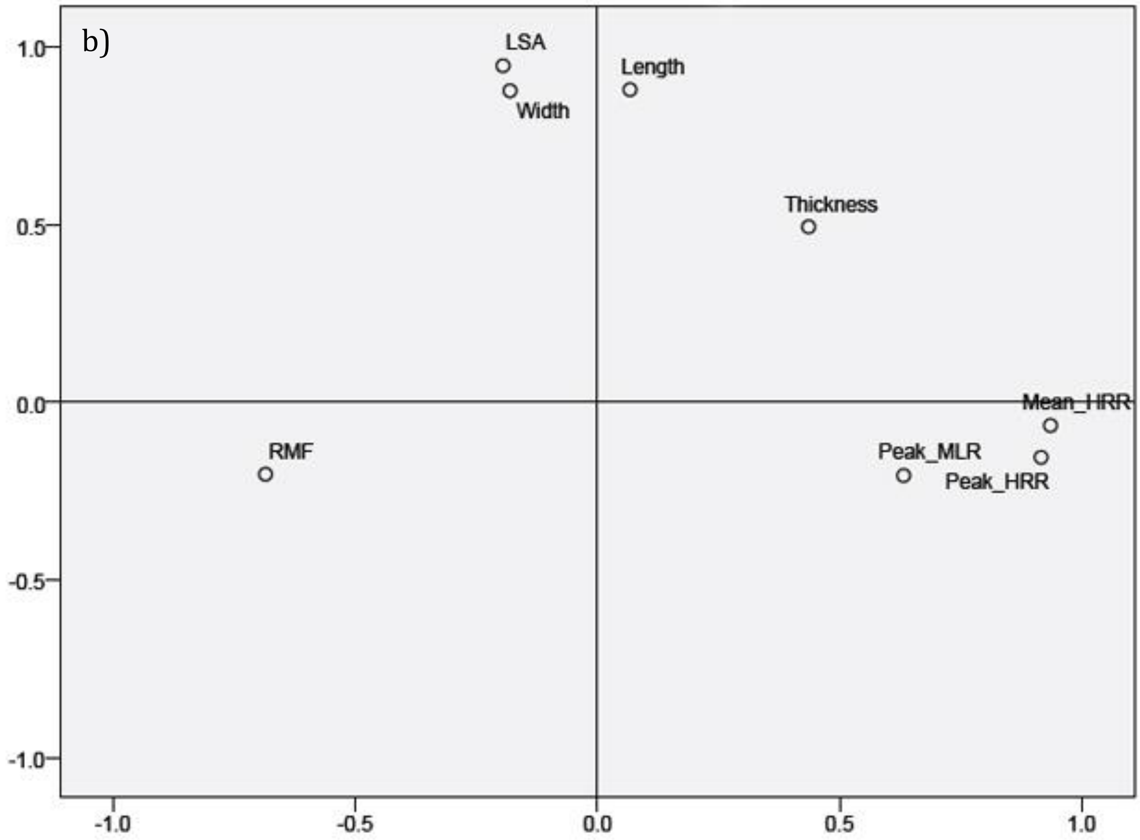

Figure 4. 5. (a) Principal component analysis loadings. LSA = leaf surface area, RMF = residual mass fraction, HRR = heat release rate, MLR = mass loss rate.

4.4.4. Plant groups and their relationships

Six groups were formed with 70% similarity level between attributes (Figure 4.6). Group 1 was composed of the invasive species *Chrysanthemoides monilifera*, *Cotoneaster coriaceus*, *Olea europea* sp. *cuspidata*, *Pyracantha* sp., *Rhamnus alaternus* and *Vinca major*. Group 2 contained only native species: *Acacia implexa*, *Acacia rubida*, *Banksia serrata*, *Myrsina variabilis* and *Smilax australis*. Group 3 was composed of a mix of natives and invasive species: *Angophora costata*, *Backhousia myrtifolia*, Bambuseae spp. and *Rubus fruticosus* agg. Group 4 was formed by the invasive species *Acacia baileyana*, *Cystisus scoparius*, *Ligustrum vulgare*, *Lycium ferocissimum* and *Senna pedula* var. *glabrata* and the native species *Acacia parramatensis*. Group 5 contained *Erythrina crista-galli* and *Solanum mauritianum*. Group 6 was formed by the invasive species *Ageratina adenophora*, *Cestrum parqui*, *Cinnamomum camphora*, *Hedera helix*, *Lantana camara*, *Ligustrum lucidum*, *Ligustrum sinense*, *Lonicera japonica*, *Pittosporum undulatum*, *Prunus cerasifera* and *Rhus typhina* and the native species, *Callicoma seratifolia*.

There were overall effects when comparing groups with the factor scores of the PCA (Table 4.7). Analysis of variance and post-hoc Tukey's test divided the six groups in three subsets when comparing them to Axis 1 of the PCA and four subsets when compared to Axis 2 of the PCA (one-way ANOVA; $P < 0.05$).

Table 4.7. Principal components analysis (PCA) factor score ± standard deviation for each group. Different letters indicate significant statistical differences among groups.

Group	PCA Factor 1	PCA Factor 2
1	0.54 ± 0.37^{ab}	-0.56 ± 0.30^{AB}
2	0.85 ± 0.42^{ab}	0.16 ± 0.18^{B}
3	1.48 ± 0.87^{a}	-0.16 ± 0.43^{C}
4	-1.08 ± 0.36^{c}	-1.18 ± 0.17^{A}
5	-1.08 ± 0.37^{c}	3.09 ± 1.22^{D}
6	-0.33 ± 0.69^{bc}	0.31 ± 0.36^{C}

Groups 1, 2 and 3 were similarly affected by PCA Factor 1 and were separated due to high values of mean HRR, peak HRR or peak MLR (Table 4.8). Groups 4, 5 and 6 were affected by RMF indicating scoring higher values for this variable.

The four subsets formed when comparing groups with PCA Factor 2 were related exclusively to leaf traits (Table 4.8). Groups 1, 2, 3 and 6 showed an even distribution of values in the middle range of values found for LSA, leaf length and leaf width. Group 4 had the smallest values of LSA, leaf length and leaf width and Group 5 included plants with large leaves.

When the 'condition' (native or invasive species) was taken in account, a significant statistical difference was found between groups in relation to PCA Factor 1 (one-way ANOVA; P = 0.003), but no significant differences were found between groups and PCA Factor 2 (one-way ANOVA; P = 0.845).

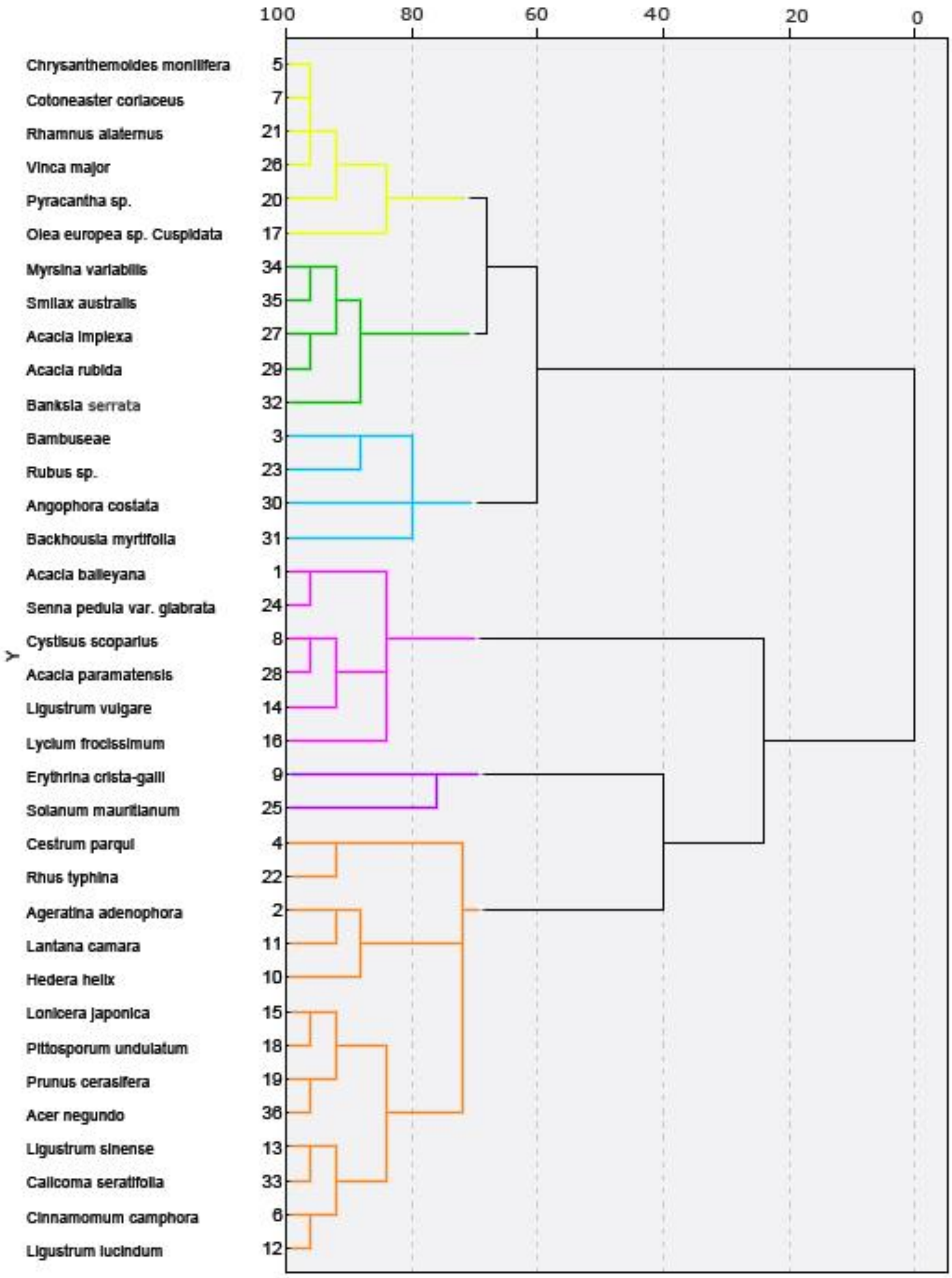

Figure 4. 6. Cluster dendogram of 36 species determined by traits composing PCA Factor 1 and PCA Factor 2. Groups were defined at 70% similarity among species.

Table 4.8. Range of values for leaf surface area, leaf thickness, leaf length, leaf width, residual mass fraction, mean heat rate release, peak mass loss rate and peak heat rate release within each group derived from cluster analysis. LSA = Leaf surface area, RMF = Residual mass fraction, HRR = Heat release rate, MLR = Mass loss rate.

Groups	LSA (cm^2)	Thickness (mm)	Length (cm)	Width (cm)	RMF (%)	Mean HRR ($kW\ m^{-2}$)	Peak MLR ($g\ s^{-1}$)	Peak HRR ($kW\ m^{-2}$)
1	1.95 - 11.09	0.24 - 0.33	3.19 - 8.22	1.25 - 3.04	0.11 - 0.17	149.36 - 171.64	0.14 - 0.22	186.07 - 260.50
2	12.26 - 16.43	0.27 - 0.44	6.23 - 12.85	1.70 - 3.17	0.03 - 0.09	139.66 - 157.09	0.15 - 0.18	175.67 - 213.11
3	12.13 - 22.94	0.12 - 0.37	5.57 - 13.42	1.98 - 3.20	0.06 - 0.09	146.18 - 219.16	0.23 - 0.34	182.70 - 282.67
4	0.58 - 6.51	0.15 - 0.35	1.37 - 4.69	0.85 - 1.95	0.14 - 0.21	92.37 - 123.73	0.10 - 0.24	146.02 - 162.74
5	91.88 - 147.64	0.22 - 0.48	13.36 - 24.42	9.68 - 11.44	0.11 - 0.16	102.28 - 112.96	0.10 - 0.12	126.23 - 145.28
6	14.54 - 49.61	0.16 - 0.35	7.59 - 14.12	2.61 - 8.30	0.08 - 0.16	93.74 - 152.92	0.10 - 0.23	116.33 - 207.51

4.4.5. Plant flammability rank

The overall flammability of 39 species was determined by ranking the four components of flammability (ignitability, sustainability, combustibility and consumability). For the top five most flammable species only one species was native (*Acacia parramattensis*). This species occupied the second position behind *Cystisus scoparius.* The last five positions were occupied by two native and three invasive species. In order for a species to be classified as having low flammability this plant had to have low scores in one or more flammability components which would set it in a lower score. *Ulex europeus* and *Lantana camara* were on the bottom of the list while the last position in the rank was the native *Smilax australis* which is a crawling vine with very thick leaves and high mineral content (Table 4.9).

Table 4.9. Flammability rank and weighted score for each flammability component of species burned using the mass loss calorimeter. N = native species, I = invasive species

Flammability rank	Species	Condition	Ignitability	sustainability	combustibility	consumability
1	*Cystisus scoparius*	I	83.55	100.00	3.05	18.55
2	*Acacia parramattensis*	N	73.62	88.86	24.00	12.33
3	*Lycium ferocissimum*	I	49.19	48.56	13.23	0.80
4	*Pyracantha* sp.	I	90.88	50.81	48.58	38.82
5	*Pinus radiata*	I	73.29	40.30	31.31	44.93
6	*Olea europea* subsp. *cuspidata*	I	84.04	36.55	50.58	20.64
7	*Ricinus communis*	I	84.69	33.29	0.00	0.00
8	*Ligustrum vulgare*	I	95.28	36.92	14.39	39.66
9	*Solanum mauritianum*	I	95.44	25.87	2.06	25.32
10	*Vinca major*	I	87.46	25.16	17.48	52.92
11	*Senna pendula* var. *glabrata*	I	87.95	22.78	61.07	27.52
12	*Pittosporum undulatum*	I	94.79	21.61	14.91	54.77
13	*Chrysanthemoides monilifera*	I	94.30	22.53	37.11	54.54
14	*Hedera helix*	I	90.39	20.15	54.30	41.16
15	*Cotoneaster coriaceus*	I	96.42	20.53	30.63	52.34
16	*Acacia implexa*	N	89.25	19.11	24.49	75.00
17	*Acacia baileyana*	I	87.62	19.11	24.00	21.33
18	*Lonicera japonica*	I	93.81	17.90	16.02	41.27
19	*Myrsine variabilis*	N	97.72	18.40	30.55	82.56
20	*Rhamnus alaternus*	I	81.76	15.19	23.19	40.80
21	*Ageratina adenophora*	I	91.86	13.52	42.51	36.18
22	*Prunus cerasifera*	I	91.21	12.77	10.13	62.19
23	*Rhus typhina*	I	98.37	16.27	1.60	53.37
24	*Acacia rubida*	I	81.76	11.68	31.59	76.38
25	*Callicoma seratifolia*	N	83.71	11.68	31.68	55.41
26	*Ligustrum lucidum*	I	89.41	12.14	27.19	60.50
27	*Angophora costata*	N	86.97	10.51	100.00	79.53
28	*Erythrina crista-galli*	I	99.19	13.27	9.45	54.61
29	*Cinnamomum camphora*	I	100.00	11.60	13.43	68.56
30	*Acer negundo*	I	62.21	5.59	13.50	42.89
31	*Rubus fruticosus* agg.	I	82.57	5.38	62.62	74.62
32	*Backhousia myrtifolia*	N	98.70	10.01	54.23	83.46
33	*Cestrum parqui*	I	99.02	9.35	4.12	28.20
34	Bambuseae spp.*	I	97.23	8.76	63.70	62.43
35	*Banksia serrata*	N	85.02	5.76	36.44	99.66
36	*Ulex europeus*	I	50.81	0.63	69.52	100.00
37	*Lantana camara*	I	91.37	0.00	33.70	33.37
38	*Ligustrum sinense*	I	97.72	22.53	17.59	50.16
39	*Smilax australis*	N	0.00	6.93	32.17	64.33

4.5 Discussion

4.5.1. Morphological, chemical and flammability leaf traits

In terms of plant flammability, leaves of native plant species had higher gross heat of combustion when burnt under controlled conditions with pure oxygen. However, this difference was not apparent for mean energy heat of combustion which represents a more realistic heat output from burning vegetation. Native species could also be separated from invasive species as they burnt more completely than invasive plants leaving less unburnt mass after pyrolysis. Native plants, at least the species used in this study, can release more energy per mass unit and are more fully combusted so that fires fed by these fuels could potentially be more intense. On the other hand, invasive species had lower HRR and MLR than natives but longer flaming periods which could impact fire severity and effects in areas covered by these plants.

One of the most innovative features of the study presented here is that different types of leaf traits – morphology, chemistry and flammability were compared as a way of understanding and classifying the potential of plants to burn. Flammability traits were then combined to rank vegetation flammability. This type of information is useful from a fire management perspective for planning, control and safety of prescribed burning.

Despite the extensive number of variables tested, there was no discernible difference in morphological leaf traits among native and invasive plants. In contrast, Murray *et al.* (2013) investigated the leaves of 52 native species and 27 exotic plants occurring in dry sclerophyll forests of New South Wales and found that leaves from exotic plants were generally wider, longer and larger in area but not thicker than leaves from native plants. It should be noted that the current study had only 10 invasive and two native species in common with the work done by Murray *et al.* (2013) and that the species tested were not restricted to woody life forms. Regardless of this, the lack of difference in morphological leaf traits among native and invasive plants is in agreement with the hypothesis that plant communities are composed of species that have analogous adaptations to a particular physical environment (Callaway 1997; Van Driesche and Bellows Jr 1996). This would allow exotic species with similar adaptations (e.g. leaf morphology) to occupy an empty niche or disturbed site and outcompete native plants with similar attributes (Callaway and Aschehoug 2000; Lake and Leishman 2004).

Leaves of invasive species had higher concentrations of most nutrients including N, K, Mg, and P indicating they have a greater ability to access and store these nutrients than native species. Funk and Vitousek (2007) showed that invasive species were more efficient at using limiting resources than native plants on short time scales and were similarly efficient when carbon assimilation per unit of resource was integrated over leaf lifespan.

Carbon fixation strategies are often regulated by leaf traits and these have a major role in the ecological strategies of plants (Wright *et al.* 2004; 2007). There are several ecologically important trade-off relations between leaf traits (Leishman *et al.* 2007). For example, specific leaf area (SLA) can be related to leaf life span (LL) where plants with low SLA normally have longer LL and therefore need more structural strength (Leishman *et al.* 2007). These species also lean towards having more resources allocated to producing chemical defences and volatiles (Coley 1988). On the other end of this spectrum are the plants with high SLA that have shorter LL, faster rates of growth and greater nutrient requirements. Leishman *et al.* (2007) found that N, P and N:P were higher for exotic plants occurring in disturbed sites. Although exotic invasive species did not have different strategies for carbon capture (i.e. high SLA, high leaf area ratio and fast relative growth ratio) compared to native plants, they were positioned further along a leaf economic spectrum towards faster growth strategies. Similarly, Baruch and Goldstein (1999) demonstrated that invasive species had greater levels of N and P in leaves coupled with lower leaf construction costs. The invasive species used in this study had higher foliar nutrient concentration than native species indicating that they have the potential to make better use of the resources available for biomass production and therefore, production of fuel.

4.5.2. Relationships among leaf traits and flammability

Plant flammability relates to how easily a plant ignites and burns. Studies have investigated leaf flammability as ignitability (Gill *et al.* 1996; De Lillis *et al.* 2009) (Saura-Mas *et al.* 2010; White and Zipperer 2010; Ganteaume *et al.* 2013; Murray *et al.* 2013), sustainability (Berry *et al.* 2011), combustibility (Dickinson and Kirkpatrick 1985; Behm *et al.* 2004), and consumability but very few studies have investigated all components of flammability together (Pausas *et al.* 2012; Madrigal *et al.* 2013). When trying to relate plant flammability to leaf foliar traits there is a lack of consistency in defining plant flammability. In addition, there is no standard way of measuring flammability traits and the few studies available only consider one or two components of flammability (Agee 1997; Dimitrakopoulos 2001; Etlinger and Beall 2005; Scarff and Westoby 2006; Schwilk and Caprio 2011; Murray *et al.* 2013). In this study, plant flammability was considered to include flammability traits including gross heat of combustion (HoC), time to ignition (TTI), flame duration (FD), residual mass fraction (RMF), mean heat rate release (Mean HRR), peak heat rate release (peak HRR), mean energy heat of combustion (mean EHC) and peak mass loss rate (peak MLR). All these traits are directly or indirectly related to flammability of plants.

As described by Gill and Moore (1996), the ignitability of fuel is defined as the ignition delay time. It is the time to first flaming combustion from the time of first exposure to an ignition source. The ignitability of fuel is often considered to be the most important flammability trait because without it there is no fire. However, ignitability is only one aspect of the complex of mechanisms involved in the process of pyrolysis. In this study there was no significant relationship between leaf morphology and ignitability. Gill and Moore (1996) found considerable variability in ignitability and attributed it to two variables – leaf moisture content and leaf surface area:volume ratio. Leaf surface area:volume ratio was also the best predictor for ignitability in the study by Atreya (1998), and is a critical factor influencing rate of spread of wildfires. Ganteaume *et al.* (2013) found a correlation between fuel moisture and time to ignition and showed that leaves with high moisture content also burned for longer (increased sustainability). In the study presented here, leaves were dried to the same moisture content to remove the confounding influence of heat yield on intra-specific variation. This would undoubtedly change time to ignition as heat is required to heat and evaporate moisture before the fuel can be ignited. Along the same lines, Ormeno *et al.* (2009) showed that different species containing different concentration of terpenes can affect the time to ignition. Drying the fuel will have removed some of the volatile organic compounds including terpenes which will also influence this aspect of flammability.

There was a strong negative correlation between flame duration and leaf length and width indicating that small leaves tended to burn for a longer time than large leaves. Schwilk and Caprio (2011) found that leaf length had an effect on the severity of fire such that vegetation with longer leaves would sustain more severe fires. Both Gill and Moore (1996) and Scarff and Westoby (2006) demonstrated that leaf size influences litter flammability, fire intensity and sustainability of fire. These results are not surprising as leaf size affects the ventilation of the fuel bed (Drysdale 2011). Small leaves can create an air-tight litter bed limiting the oxygen supply and, as a consequence, combustion reactions will be slow and the litter bed will burn for longer when compared to litter beds composed by large leaves (Scarff and Westoby 2006; Drysdale 2011). It could be hypothesised that while larger leaves tend to create open litter beds that burn better due to increased ventilation, small leaves burning at a slower rate could extend the fire residency time in the litter bed.

As a consequence of using a mass loss calorimeter requiring a sample holder of fixed volume to determine flammability traits, the packing ratio of leaf material varied across samples and species due to differences in leaf size. In the field, the packing ratio is the fraction of fuel bed volume that is occupied by fuel particles, and is a function of fuel load, fuel bed depth, and fuel particle density (Scott and Burgan 2005). The extended flame duration time for species with small leaves is likely to be due, at least in part, to the compact nature of these fuels when placed in the holder.

4.5.3. Flammability and leaf morphology as independent factors

The combustion of forest fuels is extremely complex involving multiple interrelated components and processes. A limited understanding of these physicochemical processes has restricted the development of theoretical forest fire behaviour (Madrigal *et al.* 2009). Similarly, the applicability and importance of bench-scale experiments measuring HRR and its impact in real-scale forest fires is still relatively new and remains controversial (Schemel *et al.* 2008; Madrigal *et al.* 2009;

Fernandes and Cruz 2012; Madrigal *et al.* 2013). Changing the way flammability is studied by investigating a broader range of variables associated with flammability and examining the relationships with the nature of the fuel, such as its arrangement and chemical composition, may help to overcome this limitation.

A large proportion of the variability among the species tested can be explained by the four flammability related variables: mean HRR, peak HRR, peak MLR and RMF. Species with higher mean HRR, peak HRR and peak MLR had smaller proportions of unburnt mass after pyrolysis. Most of the native species used in this study had high mean and peak HRR and peak MLR whereas invasive species had greater amounts of unburned mass after the pyrolysis and low to medium values of mean HRR, peak HRR, peak MLR. Heat release rate is considered to be one of the most important variables for charactering the 'flammability' of products (Babrauskas and Peacock 1992; Schemel *et al.* 2008). The rate of heat release influences fire characteristics such as flame geometry, temperature fields (the set of temperature values at all points in a given space at a given instant) and rates of fire propagation (Schemel *et al.* 2008). This is particularly important in cases of fire in urban environments and is likely to be equally important in modelling of fires in vegetation. Heat rate release can be used by fire modellers and managers to create predictions of the spread and intensity of fire and hazard on the basis of results obtained for a given species (Janssens 2000).

Native plants had lower values for mean and peak HRR than invasive plants. These two measures of flammability are related to combustibility such that after pyrolysis there is less unburnt mass remaining for species with high HRR. This characteristic of flammability can lead to considerably different impacts during planned and unplanned fires in pristine and weed-invaded areas because of its direct influence in the kinetic processes in the fuel bed (Janssens 2000; Schemel *et al.* 2008).

In this study, there was no discernible difference in leaf size between invasive and native species. There are only two other studies available in the published literature comparing leaf morphology of exotic species from with the local Australian flora. Both of these studies found differences in leaf size when native and exotic species were compared (Leishman *et al.* 2007; Murray *et al.* 2013). It is likely that our sample size of 39 species could have influenced the distribution of leaf size tested (samples sizes in cited references ranged from 55 to 79 species), however we cannot discard the hypothesis that woody weeds happen to have leaves distributed across a similar range of sizes found for native species, at least in some vegetation types.

The PCA indicated a clear separation among morphological and flammability traits. Leaf morphology and leaf flammability appear to be regulated by different factors and did not influenced each other for the species used in this study. Similarly, there was a clear separation between invasive and natives plants for flammability traits but not for leaf morphology traits. Future investigation involving a broader range of native and invasive plants are needed to confirm if there are enough morphological differences between the two groups and how this relate to the four components of flammability. Studies like this can help resource allocation and management of fires and invaded areas.

4.5.4. Species flammability rank and possible impacts

The importance of ranking species according to flammability is widely recognised yet extremely hard to achieve since there is as yet, no standardised way of measuring or interpreting species flammability. For example, common questions from land managers around flammability often relate to determining what garden or landscaping plants can be used near households to reduce the spread of fire or how can knowledge of the flammability of a given forest be used to reduce the risk of loss of life or property (Gill and Zylstra 2005). Similar questions are posed by the scientific community when trying to determine how the intrinsic flammability of plants affects fire behaviour (Dickinson and Kirkpatrick 1985; Dimitrakopoulos and Papaioannou 2001; de Magalhaes and Schwilk 2012) or how invasive and native plants differ in flammability (Pausas *et al.* 2012; Ganteaume *et al.* 2013; Murray *et al.* 2013). Even though there are extrinsic variables such as fuel moisture and arrangement that were not considered (Plucinski and Anderson 2008; De Lillis *et al.* 2009), this study represents the first attempt to rank species according to flammability using four components that contribute to flammability. Using a defined suite of plant species, it appears that woody weeds are putatively more flammable than native plants.

A list of species ranked according to flammability could be used to prioritise fuel hazard reduction or rehabilitation. For example, the highest ranked species in this study was Broom (*Cystisus scoparius*). It is an erect shrub to 3 m tall originally from Europe. This species is widespread and has formed major infestations in southern Australia. Invasion by Broom is promoted by disturbance but can be found spreading into native pristine vegetation (Fogarty and Facelli 1999). It is particularly difficult to restore native vegetation that is long invaded by Broom (Fogarty and Facelli 1999). The capacity of this species to colonise new areas and outcompete native plants in conjunction with its potential flammability, as demonstrated in this study, and architecture could lead to considerable changes in fire behaviour in invaded areas. Using knowledge of the biology and flammability of certain species could be used to reduce fire risk in urban areas.

Regardless of rankings derived from measures of flammability or any other system, it is important to consider that biological invasions are unique processes and each case needs to be analysed in terms of plant growth and architecture, plant establishment patterns and phenology and, in terms of fire risk, contribution to the fuel load. The species *Ulex europeus* is a good example to illustrate this point. This species is classified as highly flammable in the literature mostly due to the retention of dead branches (Pausas *et al.* 2012). However, it was at the bottom of the flammability rank list in this study. It is obvious that a wide range of traits of flammability need to be taken into account to reflect burning under natural conditions. This difference could also be attributed to the use of fresh leaves in this study considering that the composition and flammability of fresh and dead leaves are different. Despite the methodological limitations imposed by the use of a cone calorimeter, discussed in depth by Fernandez and Cruz (2012), the importance of ranking flammability using a method that allows comparisons independent of the plant origin is extremely useful. Murray *et al.* (2013) suggested that quantification of the relative input of exotic leaves into leaf litter in dry sclerophyll forest should be used alongside information provided by ranking species according to flammability. The presence of invasive species will alter the dynamics associated with turnover of leaf litter such that the accumulation of leaves from exotic species in the litter layer could lead to an increase in bushfire intensity and frequency (D'Antonio and Vitousek 1992; Brooks *et al.* 2004). Similarly, information about the architecture of the invasive plant, such as branch retention (Schwilk 2003), is essential for a more complete understanding of plant flammability.

Not all plant invasions will increase fire intensity or alter fire regimes (Mandle *et al.* 2011), however the ability to classify species according to their flammability will allows for targeted management and research.

Chapter 5. Prediction of fire behaviour in an eastern Australian woodland and a novel vegetation type

5.1. Introduction

5.1.1. Fire behaviour prediction and invasive plants

Fire behaviour can be affected by vegetation in different ways (Brooks *et al.* 2004; Watson and Wardell-Johnson 2004; Mandle *et al.* 2011). 'Intrinsic' fuel properties mostly influence fire frequency, intensity and seasonality and are related to the physiological condition of the plant (Brooks *et al*. 2004). Intrinsic properties include the moisture content and surface area of leaves, and the chemical volatility and heat content of plant tissues. These features directly affect the ignitability of fuels and the amount of heat released during combustion. 'Extrinsic' properties of fuel relate to the way plants are spatially arranged in the environment. For example, the amount of fuel per unit area, fuel continuity and the packing ratio of fuels or bulk density are considered to be extrinsic properties. These characteristics are known to affect fire intensity and frequency, and the seasonality and extent of fires (Brooks *et al*. 2004).

Many studies have inferred the effects of invasive plants on fire behaviour (D'Antonio and Vitousek 1992; Mack and D'Antonio 1998; Rossiter *et al.* 2003; Brooks *et al.* 2004; Dibble *et al.* 2007; Pauchard *et al.* 2008; Fisher *et al.* 2009; Rew and Johnson 2010; Berry *et al.* 2011). To date, there are no studies that have used field data from sites affected by invasive woody plants as inputs for fire behaviour models nor are there any studies that have compared predictions of fire behaviour between invaded and non-invaded or pristine vegetation.

One of the most widely used models for predicting fire behaviour was developed by Richard C. Rothermel (Rothermel 1972; Rothermel and Deeming 1980; Wells 2008).

The Rothermel model has been incorporated into several predictive tools such as the BehavePlus Fire Modelling System (Andrews 1986; Andrews *et al.* 2003; Andrews 2009) and the Farsite Fire Spread Simulator (Finney 1998). As such, it is currently in use by fire managers around the world (Andrews 2010).

The BehavePlus Fire Modelling System (hereafter referred to as 'the BehavePlus model'; see Chapter 1 for fuller description) is software supported by mathematical models and equations, that predicts fire behaviour (e.g. flame length and rate of spread), fire effects (e.g. scorch height and tree mortality), and the fire environment (e.g. fuel moisture and wind adjustment factor; Andrews 2009; 2013; White *et al.* 2013a). Although the BehavePlus model is a powerful tool used to predict fire behaviour it was not developed for use with Australian vegetation. To use the BehavePlus model in different fuel types, adjustments and comparison to the models currently in use need to be done. However, the only work comparing fire behaviour predictions made by a predecessor of the BehavePlus model ('Behave'; Burgan and Rothermel 1984) and the Forest Fire Danger Meter (FFDM; McArthur 1976) was done by Moore (1986). Since then, the FFDM has largely been replaced or complemented by the Vesta model (Gould *et al.* 2007a) and the BehavePlus model has been updated and improved. Regardless of these modifications and improvements, the gaps between fire behaviour prediction models remain.

5.1.2. Fire prediction in Cumberland Plain Woodland

As described in detail in Chapter 2, Cumberland Plain Woodland (CPW) is a critically endangered ecological community (Benson 1992; Benson and Howell 2002). This community is under threat from a number of anthropogenic disturbances such as clearing, logging and changing fire regimes (Benson *et al.* 1990). Invasion by exotic plants poses a significant problem for ecosystem management with most remnants of CPW containing more than five exotic plant species (Benson and Howell 2002).

In the past few decades, African Olive (*Olea europea* subsp. *cuspidata*) has become a major invasive species in CPW and is capable of forming a dense permanent mid-canopy vegetation type (Cuneo *et al.* 2009). This invasion has created a novel fuel type within CPW, possibly altering the fire behaviour in invaded areas. The BehavePlus model was used to predict fire behaviour in invaded and pristine areas of CPW and predictions were compared between the two vegetation types. In addition, results from the BehavePlus model were tested against models currently used for fire prediction in native vegetation in Australia.

5.1.3. Current fire modelling

The complexity and dynamics of fire behaviour are characterised by several different variables of which fire intensity, rate of spread and flame dimension are considered to be the main aspects (Whelan 1995). At present, fire scientists use three different approaches to predicting surface fire spread and other fire behaviour characteristics (Sullivan 2009a). The various methods used can be classified as physical and quasi-physical models; empirical and quasi-empirical models; and simulation and mathematical analog models (Sullivan 2009a; b). Each of these approaches has limitations and a physical model of fire spread that is adequate for operational fire behaviour forecasting does not yet exist (Sullivan 2009b).

Fire behaviour modelling in Australia has been based almost exclusively on an empirical approach. The first fire behaviour guides for eucalypt forest and grassland were developed by Alan McArthur (McArthur 1962; 1966; 1967) and George Peet (Peet 1965) and were based on surveys of small experimental burns (Gould *et al.* 2007a). While these models have been in use for many decades, the use of empirical models can result in misleading predictions when applied beyond the range of data that it was developed for (Gould *et al.* 2013), particularly in novel fuel types.

5.1.4. The Grassland Fire Danger Meter model

In 1966, McArthur developed the first fire spread model for grassland. Since this time, the Grassland Fire Danger Meter (GFDM) model has been used widely in Australia, including use by bodies such as the Bureau of Meteorology and most State and Territory fire authorities. This meter, together with the rate of spread and flame height predictions presented in Cheney *et al.* (1998), are the main tools used to describe the effects of weather and fuel on fire spread in grassy vegetation.

Grassfires are characterised by igniting and burning more rapidly for a given set of conditions when compared to coarser types of fuels such as forest litter (Sullivan 2013). Grass fuels present an average flaming time of approximately 5 seconds and burn-out time of 10–15 seconds. Grassfires develop very quickly from a point of ignition and burn with extremely fast speeds compared to forest fires; this allows grassfires to respond to wind changes almost instantly.

Cheney *et al.* (1993; 1998) demonstrated that fuel load has a smaller role in determining rate of spread in grassland compared to forest or woodland. The structure of fuel as described by pasture condition (i.e. continuous/discontinuous or standing/eaten out) is more important for determining forward rate of spread. Similarly, flame length is not as important during grassfires as it is in forest fires

(Cheney and Sullivan 2008). However, flame length, together with wind speed, combine to keep the fire moving forward and overcome possible gaps in the fuel (Burrows *et al.* 2009). The more discontinuous the fuel, the higher the threshold wind speed required to drive a fire forward so that flames can bridge the gaps in the vegetation (Bradstock and Gill 1993; Cheney and Sullivan 2008; Burrows *et al.* 2009; Sullivan 2013). For grassfires, fuel continuity, degree of curing, state of the fuel (i.e. grazed or ungrazed), slope and wind all have an important role in determining rate of spread and flame length (Cheney and Sullivan 2008). In this study a combination of McArthur and Cheney's works is used to predict the fire behaviour of Intermediate Invasion (II) areas (See details in Section 5.2.2).

5.1.5. The Forest Fire Danger Meter and the Vesta model

One of the central premises for the Forest Fire Danger Meter (FFDM) is that the rate of spread of fire increases with increasing fuel load (McArthur 1962; Peet 1965; McArthur 1967a). Although McArthur's pioneering work described relationships between fuel load, rate of spread and fire intensity, subsequent studies have shown that his models under-predict rate of spread and fire intensity under severe burning conditions (Rawson *et al.* 1983; Burrows 1994; 1999a; Gould *et al.* 2007a; Gould *et al.* 2013). The FFDM was primarily developed using low intensity experimental fires and some wildfire data for fire danger rating, based on difficulty of suppression. However it has been extrapolated to predict fully developed fires (Gould *et al.* 2013). For example, Burrows (1994; 1999a) and McCaw *et al*. (2008) showed that the FFDM regularly under-predicts the rate of spread of fires under dry summer conditions by a factor of two or more. More recently, McCaw *et al.* (2012) analysed the effects of fuel characteristics on fire spread for bigger fires and under severe weather conditions. This work supported conclusions that: (1) the dependence of fire rate of spread on surface fuel load is not as strong as assumed by the FFDM (McArthur 1967) and Forest Fire Behaviour Tables for Western Australia (FFBT, Sneeuwjagt *et al.* 1979); (2) the near surface fuel layer has the greatest contribution to rate of spread; and (3) it would be worth including visual hazard scores representing the quantity and arrangement of fuel into algorithms to predict fire behaviour.

The Vesta model is a system developed by Gould *et al.* (2011), which uses a technique for assessing forest fuels based on hazard ratings for distinct layers within the overall fuel complex. As a consequence, this model is thought to be more efficient and reliable than the FFDM and FFBT. The Vesta model (Gould *et al.* 2007a; b) takes into account the near surface fuel layer as this is the principal layer responsible for determining fire rate of spread.

Flame height is difficult to estimate but can be related to head fire rate of spread and elevated fuel height (Gould *et al.* 2007a). Consideration of this relationship is the main difference between predictions made by the Vesta model and the BehavePlus model. In the BehavePlus model, the elevated fuel layer is not taken in account when calculating flame height, possibly causing smaller flame height predictions. For dry eucalypt forest, the Vesta model predicts flame height reasonably well and assumes that when flame height exceeds 8 m there is the likelihood of torching or crown fires in the intermediate and overstorey canopies depending on the bark hazard and the bulk density of the vegetation (Gould *et al.* 2007a). Similar mechanisms of fire propagation are not included in the BehavePlus model.

In Australia, the fuel physical model approach used in the BehavePlus model is not widely adopted particularly after fuel validation studies have shown that the

Rothermel model does not predict fire behaviour well in grass and eucalypt litter scenarios due to their non-homogeneity of fuel (Gould 1991; Burrows 1999a; Gould *et al.* 2007a; Gould *et al.* 2011). The rate of spread of the fire front in Australian fuels seems to be more influenced by structural factors, composition and fuel continuity than only by fuel load (Cheney *et al.* 1992; Burrows 1994; 1999a; McCaw *et al.* 2008; Gould *et al.* 2011). Project Vesta developed from the need to discover variables that are more satisfactory to describe fire behaviour in eucalypt forest other than only fuel load (Gould *et al.* 2007a).

5.1.6. Predicting fire behaviour in novel fuel types

There is a constant need to improve our understanding of forest fuel and how it determines fire behaviour especially under severe weather conditions (Gould *et al.* 2007a). Although fire scientists in Australia aim to constantly improve the knowledge base for models of fire behaviour in widespread vegetation types such as eucalypt forest and grassland, creating empirical models of fire behaviour in novel vegetation types, for example, altered fuel structure and load due to weed invasion, is difficult to achieve. Building an empirical model requires burning large areas with the specific fuel type under a variety of weather and climatic conditions.

There are usually limited options for modelling fire behaviour in novel fuel types without an extensive empirical burning experiment. However, predicting fire behaviour is possible through the establishment of scenarios. White *et al.* (2013a) report the possibility of using fire simulations to explore alternative scenarios with manipulation of fuels, stand structure and weather conditions. These scenarios are based on values acquired in real situations using statistical parameters that allow calculation of averages of the vegetation and fuel load and, when combined with weather and topography data, provide an output capable of describing fire behaviour for a specific situation (White *et al.* 2013a; b). Such simulations can be used in many fire management operations including prediction of the behaviour of a fire in progress, guiding the application of back burning fires, estimation of the dangers linked to a possible fire in given vegetation type and to train firefighters.

Despite the difficulties identified in building empirical models for novel vegetation types, the development and analysis of fire behaviour scenarios using field data has great potential. The study described here therefore aims to:

(1) define the parameters required for modelling fuel in invaded and non-invaded vegetation occurring at the Australian Botanical Garden, Mt Annan;

(2) simulate fire behaviour in these vegetation types using the BehavePlus Fire Modelling System;

(3) compare the results to existing models appropriate for each vegetation type currently in use by Australian authorities (e.g. GFDM, FFDM, Vesta model).

Data presented in Chapter 2 will be used to address these aims.

5.2. Material and methods

5.2.1. Study area

The general study area is located in the Cumberland Plain region west of Sydney, Australia. The study was conducted in the Australian Botanic Garden, Mount Annan in areas representing a gradient of three stages of invasion with contrasting fuel structure and complexity (See Chapter 2). Areas of Cumberland Plain Woodland (CPW) were selected to represent a pristine non-invaded environment. The other extreme of the invasion gradient was represented by areas classified as 'Long-term invasion by African Olive' (LI). These areas have been invaded by African Olive for about 15–20 years and present an advanced stage of environmental degradation (Cuneo and Leishman 2006). Between the extremes of the invasion gradient used, areas were identified and classified as 'Intermediate invasion' (II). The II areas are former pasture that were originally CPW and have been invaded by African Olive in the past two decades (See Chapter 2).

5.2.2. Model descriptions

BehavePlus Fire Modelling System

The BehavePlus model (version 5.0) is made up of over 40 deterministic mathematical models and consists not only of the models, but also model linkages and the user interface (Andrews 2013). In this study it will be referred to as a model considering the equations and algorithms that form the basis of the fire modelling system. The BehavePlus model is composed of mathematical models that are grouped into modules based on rules and assumptions and is organised in such a way that each module contains related mathematical models that can be used independently or linked together (Andrews 2013).

In this study only the 'SURFACE' module of the BehavePlus model was used. The principal model of the SURFACE module is the Rothermel Surface Fire Spread Model (1972; see Chapter 1) incorporating adjustments by Albini (1976). The Rothermel model calculates the rate of fire spread independent of source of ignition considering that the fire front in the flaming zone is predominantly influenced by fine fuels. Fire intensity and flame length calculations are based on models developed by Byram (1959) linked to Albini's and Rothermel models. The calculation of rate of surface fire spread and intensity requires a description of the surface fuel, midflame wind speed, slope and fuel moisture.

According to Andrews (2013), the surface fuel can be input to the SURFACE module in many ways: (1) as standard fire behaviour fuel models; (2) as fuel parameters; and (3) as custom fuel models, (4) two fuel models, and (5) special case fuel models. In this study, three custom fuel models were created based on vegetation characterisation (see Chapter 2). Each of these models were specified by providing the BehavePlus model with parameter inputs: fuel load (kg m^{-2}), fuel bed depth (m), bulk density (kg m^{-3}), 1-h dead and live fuel moisture (%), surface area to volume ratio (SA/V, m^{-1}) and fuel heat content (kJ kg^{-1}). Once the fuel models were constructed they were tested by simulating fires in each vegetation type under the same environmental conditions (i.e. wind speed, fuel moisture and slope).

Forest Fire Danger Meter

For the Forest Fire Danger Meter (FFDM), rate of spread of the headfire (R) is directly proportional to the load of fine fuel (<6 mm diameter) consumed (w) and is expressed in a linear relationship:

$$R = Fw \qquad [5.1]$$

where F = Forest Fire Danger Index (FFDI; McArthur 1967).

The FFDI can be calculated according to Noble *et al*. (1980) as:

$$F = 2.0\exp(-0.450 + 0.987 * \ln(D) - 0.0345 * H + 0.0338 * T + 0.0234 * V) \qquad [5.2]$$

where F = fire danger index

D = drought factor

T = air temperature (°C)

H = relative humidity (%)

V = average wind velocity (km h^{-1}) in the open at a height of 10 m.

The Vesta model

The Vesta model gives special importance to the structure of the fuel complex by combining a hazard rating for each of the different fuel layers i.e. canopy, bark, elevated (shrub fuels), near-surface, and surface fuels (see Chapter 2). The fuel hazard is rated using categorical scores from 0 to 4 based on visual assessment for the percent cover score (PCS) and the fuel hazard score (FHS) for each of the five fuel layers, following the concept of Cheney *et al*. (1992), Wilson (1992) and Tolhurst *et al*. (1996). After the vegetation is given a hazard score, these numbers can be used in conjunction with a fire behaviour table (Gould *et al.* 2007b). The tables are derived from mathematical equations relating fire behaviour characteristics (rate of spread (ROS), flame height) to wind, fuel structure, fuel moisture and slope for different hazard ratings. The complete description of the prediction system is given in Gould *et al.* (2007a).

The Grassland Fire Danger Meter

The Grassland Fire Danger Meter (GFDM) model consists of a set of tables based on measurement of experimental fires. McArthur's first model was altered and developed and the mathematical background explaining the fire spread equations can be found in Noble's work (Noble *et al.* 1980). Cheney *et al.* (1998) model replaced the McArthur's equations to better take into account fuel structure. The rate of spread in grasslands fire spread model developed by Cheney *et al.* (1998) depends on the initial growth of the fire, the pasture type, wind speed, and live and dead fuel moisture. The general model to predict rate of fire spread after the fire completed its growth phase can be written as:

$$R_{ss} = f\,(I, U_{10}, M_f, C) \qquad [5.3]$$

where R_{ss} = potential quasi-steady rate of spread (units)

I = pasture type

U_{10} = mean wind speed at 10 m in the open (the standard exposure for wind measures in Australia (km h^{-1}))

M_f = moisture content of grass (%)

C = curing stage of the grass (%).

Each of the variables are combined in a function described in Cheney *et al.* (1998). These equations were used to calculate rate of fire spread in the II areas used in this study.

5.2.3 Sampling procedures for populating fire behaviour models

In each of the three types of vegetation (CPW, II, LI), three permanent plots of 50 × 50 m were established to describe the fuel complex (see Section 2.2.2.). Three parallel transects of 50 m were established in each plot; two transects located 5 m away from the edges of the plot and one transect through the middle (i.e. at 25 m) were established within plots and data was collected along these transects to determine fuel type, arrangement and distribution using a number of methods.

Visual Hazard Scoring System

Visual scoring of fuel (see Section 2.2.3) used an adaptation of the system described by Cruz *et al.* (2010). This method allows numerical characterisation of the various fuel layers by visually estimating cover and hazard scores ranging from 0 to 4 (Gould *et al.* 2011).

Fuel loads and fuel bed depth

Destructive sampling was used to estimate fuel load, arrangement, bulk density and proportion of dead and live fuels for each fuel layer (i.e. litter, near-surface, elevated). Within each 50 × 50 m plot, two 1 × 1 m subplots were randomly located along the three sampling transects, total six sub-plots (see Section 2.2.3). Data collected for biomass determination was re-worked into time lag fuels as used in United States (Fosberg 1970).

The 10-h fuels were determined by counting all twigs (0.6–2.5 cm diameter) intercepted by the transects established in the 50 × 50 m plots. The 10-h total biomass was calculated using Brown's Woody Material formula (Brown 1974):

$$W = ((\pi^2/8 * n * QMD^2 * P_p)/L \qquad [5.4]$$

where W = fuel load (T ha^{-1})

n = number of points of representing intersecting fuels

QMD = quadratic mean diameter (cm)

P_p = wood density (g cm^{-3}; data obtained from published literature for African Olive and measured for eucalypts (see below))

L = length of transect (m).

Once fuels had been collected, categorised and weighed, representative bulk samples taken from the litter layer, dead and live 'low' fuels (0–50 cm), and dead and live 'high' fuels (50–200 cm) were placed in paper bags and fresh weights recorded. Samples were oven-dried at 105 °C for 48–72 h until constant weight was reached to determine the fuel moisture content. The FMC was used to convert measured biomass to dry-mass. Litter depth was assessed at the same observation points used for visual scoring by measuring litter-bed height according to the method described by McCarthy *et al.* (1999).

The average mass of 100-h fuels was calculated by measuring all the coarse woody debris (CWD) along the three transects in each 50 × 50 m plot. All CWD >25 mm diameter intersected by the three 50 m transects were recorded for diameter, length and state of decay and the mass was calculated according to the methodology used by Baker and Chao (2009).

Dead fuel moisture of extinction

Experimental ignition studies were conducted in the laboratory using an adaptation of the methodologies used by Plucinski and Anderson (2008) and Ganteaume *et al.* (2009). Reconstructed litter beds made with litter of uniform moisture and bulk density were built using litter collected from CPW and LI sites at Mt Annan. Fuel moisture content was used as a covariate throughout these experiments as it is an input required for the BehavePlus model.

Litter collected from each vegetation type (CPW and LI) was oven-dried at 105 °C for 48–72 h until constant weight was reached. Subsamples of dried litter (n = 10 replicates per treatment) were weighed (approximately 5 g) and allocated to different moisture treatments (0, 10, 20 and 30% of litter dry weight). Samples were prepared by adding the required amount of water to the samples in sealed containers and allowing them to come to equilibrium overnight. The litter samples were spread evenly in a round heat-proof aluminium tray (diameter of 25.0 cm and a depth of 7.5 cm) and fuel depth was measured at five locations within the tray. To facilitate ignition, a cotton ball injected with 2 ml of methylated spirits was placed in the centre of the tray and ignited using a hand-held lighter (BIC® Multi-purpose lighter, BIC USA Inc., Shelton, US). For each burn the following information was recorded: percentage of successful ignition of the fuel and percentage of times the fire successfully reached the edges of the tray. These variables allowed estimation of the fuel moisture of extinction to be used as an input for the fire behaviour module in the BehavePlus model.

Midflame wind speed and wind adjustment factor

The 10 m wind speed for three days (11, 12 and 13 of September 2013) was obtained from the Australian Bureau of Meteorology for the closest weather station

(Campbelltown, 409 m from the CPW site, 2125 m from the LI site) and compared with data acquired on the same dates from a portable weather station (WH-2080, Fine-Offset Co, Shenzhen, China) installed in CPW and LI at 2m height and under the crown. The data was averaged every 30 min and the ratio was calculated to be used as the wind adjustment factor in the BehavePlus model.

Fuel heat content and fuel surface area to volume ratio

There is considerable variation in heat content (*H*) within similar fuel types (e.g. Williams *et al*. 1998), as well as between plant species (e.g. Gillon *et al*. 1997). In instances where specific values of *H* are not known, values of 15.5 MJ kg^{-1} (Griffin and Friedel 1984) have been used for savannah-like vegetation, and values of 18.7 MJ kg^{-1} (Alexander 1982) or 21.4 MJ kg^{-1} (Susott 1982) have been used for forests. This study used the values adopted by Alexander (1982) for the models applied to predict fire spread in CPW and II (18.7 MJ kg^{-1}). The value of *H* (20.2 MJ kg^{-1}) used for the models applied to predict fire behaviour in LI areas was obtained by burning a bulked sample of green leaves (oven-dried at 75 ˚C for 48 h and finely ground; see Section 4.2.2) in an oxygen bomb calorimeter (Parr 6400 Calorimeter, Illinois, USA).

The 1-h surface area to volume ratio (SA/V) is a fuel model parameter used as an input for the fuel models used by the BehavePlus model. The 1-h SA/V is the amount of area on the outside of the fuel (surface area) divided by the volume of the fuel. The 1-h fuel is dead fuel <6 mm diameter. Input values for the 1-h SA/V were obtained from the literature using the Rothermel (1983) comparison methodology, improved by Scott and Burgan (2005).

5.3.4. Fire behaviour simulations

The fire behaviour for each vegetation type (CPW, II and LI) was modelled with the most appropriate models available (Table 5.1). The range of wind speed was varied from 0–50 km h^{-1} allowing comparisons across all the models used for each vegetation type. Grass curing of 100% and a Drought Factor of 10 was used to represent worst case conditions. The simulated range for dead fuel moisture (DFM) was 5–20%.

Table 5.1. Models used to predict fire behaviour for each of the study areas.

Cumberland Plain Woodland	Intermediate invasion	Long-term African Olive invasion
BehavePlus (Andrews 1986), Forest Fire Danger Meter (McArthur 1976), Vesta (Gould *et al.* 2007b)	BehavePlus (Andrews 1986), Grassland Fire Spread Model (Cheney *et al.* 1998)	BehavePlus (Andrews 1986)

Simulations using the BehavePlus model

After the fuel model was built for each vegetation type, the data was inserted in the BehavePlus model in order to simulate: (1) maximum surface rate of spread; (2) heat release per unit area; (3) fireline intensity; and (4) flame length. Andrews (2009) report the Rothermel Fire Spread model calculations as highly sensitive to the fuel bed depth therefore the fuel bed depth was assumed to be homogeneous. The fuel bed depth measured in the field for LI was 0.02 m. A sensitivity analysis was performed to quantify the contributions of fuel bed depth for fire predictions in LI. As a baseline for this analysis, a reference scenario was used with mean parameter values from the LI area. The predictions were then compared to the data collected from the prescribed burn applied to one of the study plots to confirm the reliability of the model and determine which one best fits to the reality observed in the field. A more comprehensive data set involving experimental burning of more than one area would be desirable for future work to make predictions more robust. For II and CPW, surface fuel height was used as fuel bed depth. Inputs for the fuel models created for the BehavePlus model are presented in Table 5.2.

Table 5. 2. Input data used in the construction of fuel models for areas of long-term invasion with African Olive (LI), intermediate invasion (II) and Cumberland Plain Woodland (CPW) using the BehavePlus model.

Input variables	Input value(s)		
	LI	II	CPW
Fuel/vegetation, surface/understorey			
Fuel load transfer portion	0	0	0
Fuel model type	D	D	D
1-h fuel load (t ha^{-1})	7.2	3.6	4.8
10-h fuel load (t ha^{-1})	8.22	0.7	4.3
100-h fuel load (t ha^{-1})	12.83	0	18.8
Live herbaceous fuel load (t ha^{-1})	0	1.5	0.2
Live woody fuel load (t ha^{-1})	0	0	0
1-h SA/V (m^2 m^{-3})	5906	7218	5906
Live herbaceous SA/V (m^2 m^{-3})	5249	6562	5249
Live woody SA/V (m^2 m^{-3})	5249	4921	4593
Fuel bed depth (m)	0.02, 0.06, 0.19	0.36	0.24
Dead fuel moisture of extinction (%)	20	40	30
Dead fuel heat content (kJ kg^{-1})	20211	18622	18622
Live fuel heat content (kJ kg^{-1})	20211	18622	18622
Fuel moisture			
Dead fuel moisture (%)	5, 10, 15, 20	5, 10, 15, 20	5, 10, 15, 20
Live fuel moisture (%)	100	100	100
Foliar moisture (%)	100	100	100
Wind adjustment factor	0.1	1	0.3

Simulations using the Vesta and Forest Fire Danger Meter models

The fuel characterisation data was used together with the Vesta tables (Gould *et al.* 2007b; 2011) and the Forest Fire Danger Meter (McArthur 1967) to calculate fire rate of spread and flame height curves which were then compared with outputs from the BehavePlus model and the limited data from a prescribed fire applied to the study site.

To calculate the Vesta model predictions, FHS and PCS values for the surface and near-surface fuels were combined and compared to fire behaviour tables presented in Gould *et al.* (2007b). The model predicts rate of spread of a fully developed fire in dry eucalypt forest with a shrub understorey and should applicable to any eucalypt fuel that is dominated by leaf litter and native shrubs with only a relatively small fraction of grass in the understorey. The predicted rate of spread will be the potential quasi-steady rate of spread for a fire burning under summer conditions after the fire has undergone its initial growth phase and each curve is adjusted according to the representative fuel moisture (Gould *et al.* 2007a).

To predict fire behaviour using the Forest Fire Danger Index (FFDI; McArthur 1967), indices had to be calculated for each dead fuel moisture condition under different wind speed scenarios. The FFDI was calculated as a function of fuel moisture, wind and fuel according to equations presented by Matthews (2010). The flame height obtained in the predictions was transformed to flame length using the equations provided by Albini (1981).

Grassland Fire Danger model

As the main fuel in II was grass, the rate of fire spread and flame height predictions were calculated using the vegetation characterisation data as inputs for the GFDM model for native ungrazed grass (Cheney *et al.* 1998). When using the GFDM it was necessary to transform the data from flame height to flame length using the equations provided by Albini (1981).

5.3. Results

Fire behaviour predictions for CPW varied according to the dead fuel moisture scenario and wind speed. To simplify the presentation of the data, the results described here are those using maximum values in a 50 km h^{-1} wind speed at 10 m in the open wind speed scenario and any variations are a consequence of a change in DFM. Even though the BehavePlus model and FFDI can predict rate of spread and flame length for wind speeds above 50 km h^{-1}, the results presented here are shown only to this speed limit to allow comparisons with the Vesta model.

Dead fuel moisture of extinction

The dead fuel moisture of extinction was determined prior to fire simulations (Table 5.3). For the fuel model for LI, the moisture content of extinction was set at 20% (Table 5.2).

Table 5. 3. Ignition success rate and percentage of times the fuel bed burned to the edge of the tray for different fuel moisture content (0, 10, 20 and 30%)for surface fuel from areas of long-term invasion with African Olive (LI) and Cumberland Plain Woodland (CPW).

Fuel moisture (%)	Ignition success rate (%)		Times burned to edge (%)	
	LI	CPW	LI	CPW
0	100	100	100	100
10	100	100	20	100
20	100	100	0	100
30	70	90	0	70

5.3.1. Predicted fire behaviour in Cumberland Plain Woodland

Rate of spread

When using the Vesta model it was predicted that the ROS using a scenario of 50 km h^{-1} wind speed at 10 m in the open varied as a consequence of changes in DFM (Figure 5.1.a.). For the 5% DFM scenario, the maximum predicted rate of spread was 30.3 m min^{-1} and this speed dropped as DFM increased. For 10% DFM, the maximum predicted rate of spread was 10.8 m min^{-1}, at 15% DFM was 5.8 m min^{-1}, and using a 20% DFM scenario was 3.8 m min^{-1}. Although the fuel moisture of extinction for the CPW was found to be 25%, the Vesta model does not make predictions above 20% DFM. Only the predictions made to a maximum of 20% DFM are presented to allow comparisons with other models.

The FFDM showed that for the 5% DFM scenario, the maximum ROS for wind speeds of 50 km h^{-1} was 3.6 m min^{-1}. For the same wind speed, ROS decreased as DFM increased and at the higher end of the dead fuel moisture content (20%) the predicted ROS was 0.1 m min^{-1} (Fig. 5.1b).

The ROS predictions from the BehavePlus model for 5% DFM using the 50 km h^{-1} wind speed scenario was 4.1 m min^{-1}. For 10% DFM, the ROS decreased to 3.0 m min^{-1}. At 15% DFM the ROS was 2.5 m min^{-1} and at 20% DFM was 2.1 m min^{-1} (Fig. 5.1c).

The predicted ROS was faster for lower DFM conditions for all models. In the 5% DFM scenario at a 50 km h^{-1} wind speed, the ROS was slowest for the FFDM model (3.6 m min^{-1}), only slightly faster for the BehavePlus model (4.1 m min^{-1}) and 10-fold faster for the Vesta model (30.3 m min^{-1}; Table 5.4). At the high end of the fuel moisture scenario, the pattern was similar with the slowest predicted ROS for the FFDM model (0.1 m min^{-1}), and the highest predicted by the Vesta model (3.8 m min^{-1}).

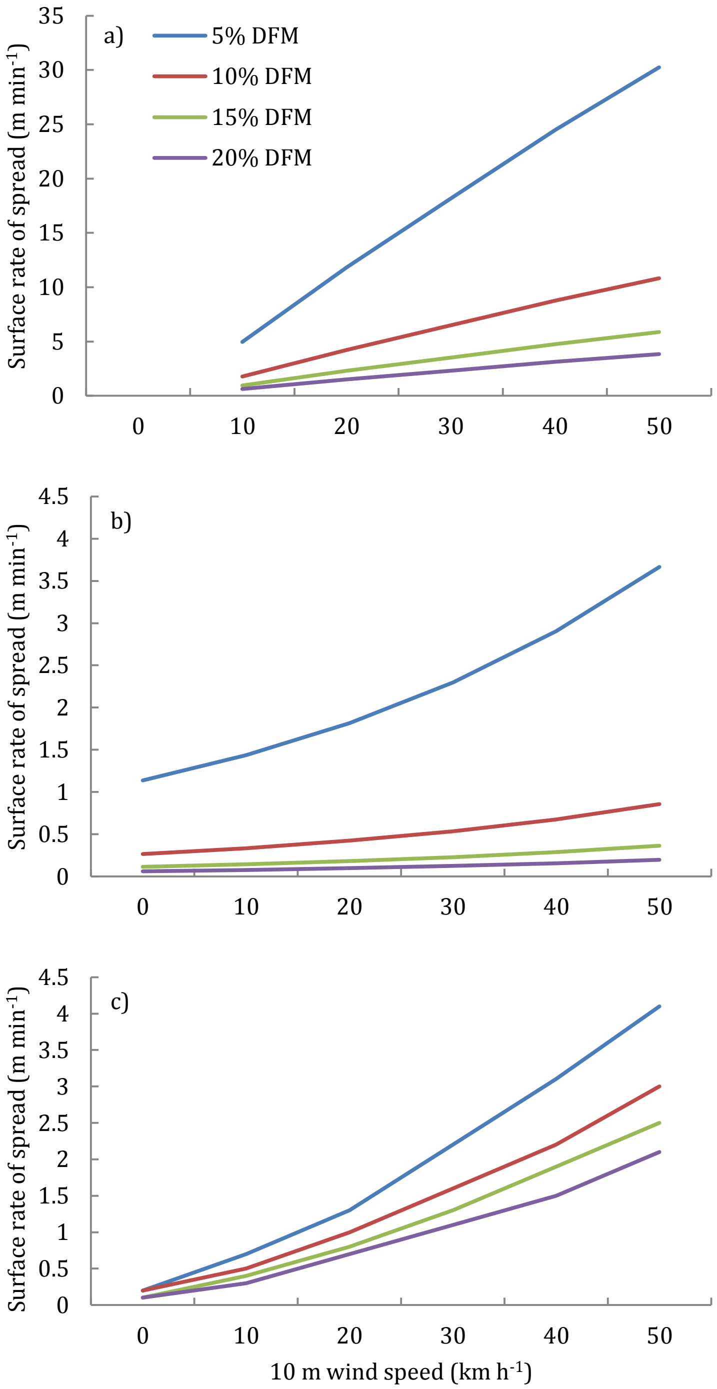

Figure 5. 1. Predicted surface rate of forward spread for Cumberland Plain Woodland by three different models: (a) the Vesta model; (b) the Forest Fire Danger Meter; and (c) the BehavePlus model at 5, 10, 15 and 20% dead fuel moisture (DFM) conditions under increasing wind speed conditions.

Table 5. 4. Rate of spread and flame length predictions for Cumberland Plain Woodland using a 50 km h^{-1} wind speed scenario by BehavePlus, Vesta and Forest Fire Danger Meter (FFDM) models under different fuel moisture conditions.

Fuel moisture (%)	Rate of spread (m min^{-1})			Flame length (m)		
	BehavePlus	Vesta	FFDM	BehavePlus	Vesta	FFDM
5	4.1	30.3	3.6	1.7	12.4	9.1
10	3	10.8	0.8	1.1	6.6	4.9
15	2.5	5.8	0.3	1	4.6	3.8
20	2.1	3.8	0.1	0.9	3.7	3.4

Flame length

The flame length/height calculations used by each model differed greatly. When using the Vesta model, flame height prediction is directly related to the rate of spread of the fire head and the elevated fuel height (Gould *et al.* 2008). The mean elevated fuel height measured in CPW was 1.87 ± 0.07 m. The maximum flame length predicted for 5% DFM and 50 km h^{-1} wind speeds was 12.4 m and was found to decrease with increasing DFM (Table 5.4). Consequently, maximum flame length predicted using the same wind speed scenario was 6.6, 4.6 and 3.7 m for 10, 15 and 20% DFM, respectively (Fig. 5.2a).

When using the FFDM model, the flame height was firstly calculated according to the equations provided by Noble *et al*. (1980) however these did not represent the original tables on the meter in a satisfactory way. A linear equation based on the FFDM tables (McArthur 1967) was instead calculated to offer a better model for flame height:

$$H = 10.698(R/1000) + 0.2656 \quad [5.5]$$

where H = flame height (m)

R = rate of spread (m h^{-1}).

This equation only applies for a fuel load of 4.8 t ha^{-1}. Flame height was then transformed to flame length using the equations provided by Albini (1981) allowing comparisons with the results produced by the BehavePlus and Vesta models.

The maximum flame length predicted by the FFDM model in a 5% DFM scenario at 50 km h^{-1} wind speeds was 9.1 m. The maximum flame length predicted for the same wind speed scenario was 4.9, 3.8 and 3.4 m for 10, 15 and 20% DFM, respectively (Fig. 5.2b).

Flame length prediction from the BehavePlus model for 5% DFM using the 50 km h^{-1} wind speed scenario was 1.7 m. For 10% DFM, the flame length decreased to 1.0 m. At 15% DFM, the flame length was 1.0 m and was 0.9 m at 20% DFM (Fig. 5.2c).

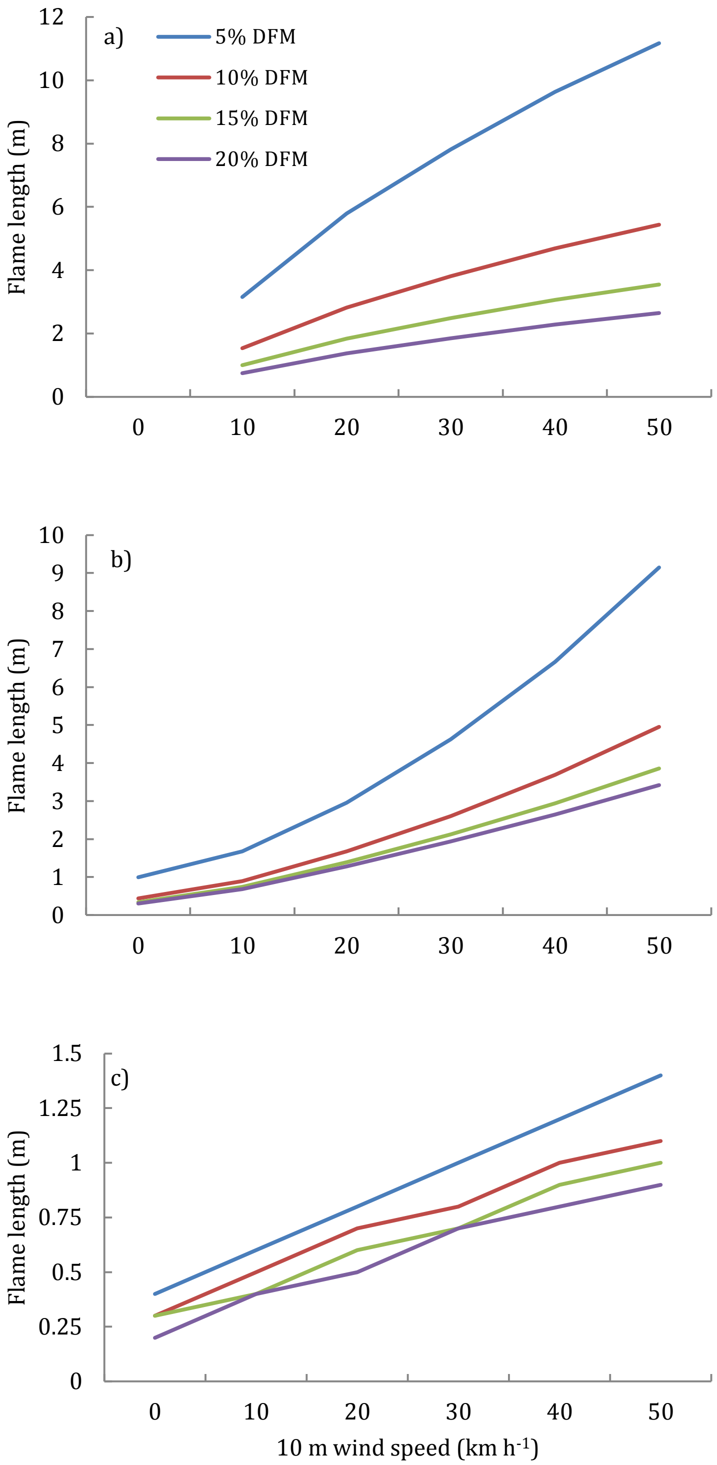

Figure 5. 2. Predicted flame length for Cumberland Plain Woodland by three different models: (a) the Vesta model; (b) the Forest Fire Danger Meter; and (c) the BehavePlus model at 5, 10, 15 and 20% dead fuel moisture (DFM) conditions under increasing wind speed conditions.

Fireline intensity

Only the BehavePlus model is able to predict fireline intensity. For 5% DFM and a wind speed of 50 km h^{-1}, the fire line intensity was 540 kW m^{-1}. For 10% DFM, the intensity declined to 325 kW m^{-1}. The predicted fire line intensity for 15% DFM was 257 kW m^{-1} and at 20% DFM the fire line intensity was 198 kW m^{-1}. The same parameter in a 25% DFM scenario was predicted to be 92 kW m^{-1} (Fig. 5.3).

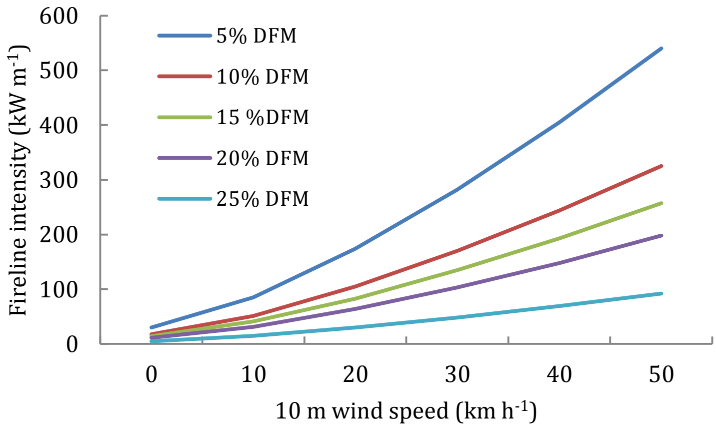

Figure 5. 3. Fireline intensity predictions for Cumberland Plain Woodland from the BehavePlus model using different dead fuel moisture (DFM) conditions under increasing wind speed conditions.

5.3.2. Predicted fire behaviour in areas of intermediate invasion

Rate of spread

When using the BehavePlus model, the surface ROS with a 50 km h^{-1} wind speed scenario was predicted to vary as a consequence of changes in DFM up to 35% moisture. Even though the BehavePlus model can predict ROS and flame length for wind speeds above 50 km h^{-1} and DFM greater than 20%, the results presented in this work are shown only to a maximum of 50 km h^{-1} wind speed to allow comparisons between models. For 5% DFM, the maximum predicted ROS was 60.3 m min^{-1} (Fig. 5.4a). For

10% DFM, the maximum predicted ROS was 31.0 m min^{-1}, 21.1 m min^{-1} at 15% DFM and 17.0 m min^{-1} at 20% DFM. It is important to highlight that the ROS in the BehavePlus model includes a wind limit function that is based on the assumption that higher intensity fires can withstand higher wind speeds than fires with lower intensities. This is particularly important for grass fires because a small wind reduction factor means fires are more likely to reach the limit than forest fires (Andrews *et al.* 2013).

The GFDM model predicted that using a 5% DFM scenario, the maximum ROS for wind speeds of 50 km h^{-1} was 105.3 m min^{-1}. For 10% DFM under the same wind conditions, the rate of spread was almost halved to 61.3 m min^{-1}, and kept decreasing as DFM increased. The predicted rate of spread at 15 and 20% DFM was 37.0 and 16.4 m min^{-1}, respectively (Fig. 5.4b). The GFDM model predicted a higher ROS than the BehavePlus model except at high moisture contents (Table 5.5).

Table 5. 5. Rate of spread and flame length predictions for a 50 km h^{-1} wind speed scenario from the BehavePlus model and the Grassland Fire Danger Meter (GFDM) model under different fuel moisture conditions in areas of intermediate invasion.

Fuel moisture (%)	Rate of spread (m min^{-1})		Flame length (m)	
	BehavePlus	GFDM	BehavePlus	GFDM
5	60.3	105.3	2.5	4.2
10	31.0	61.3	1.7	3.7
15	21.1	37.0	1.3	3.3
20	17.0	16.4	1.2	2.8

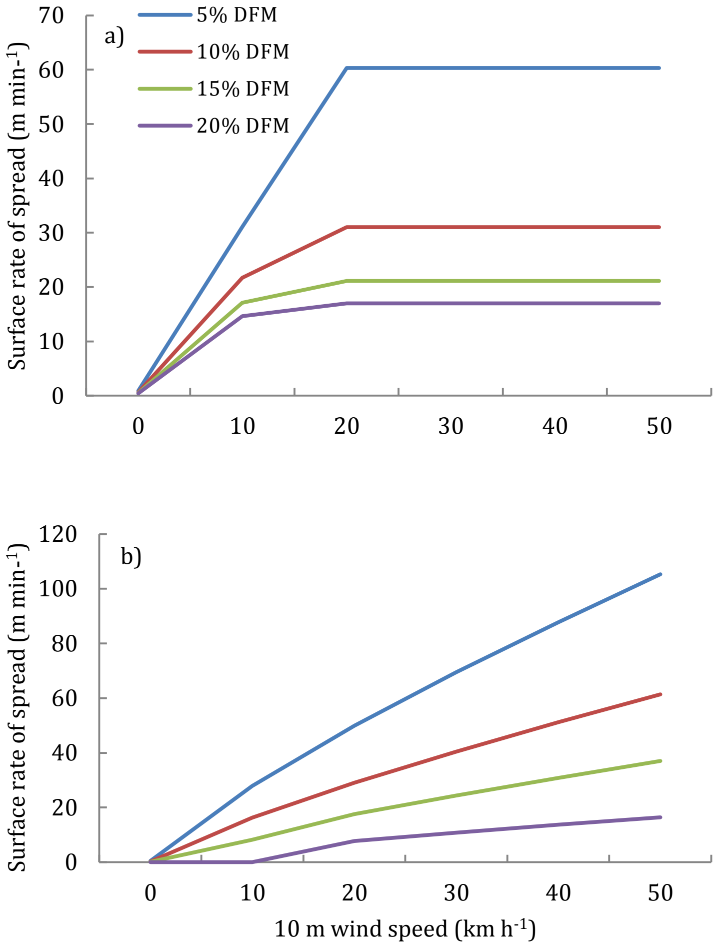

Figure 5. 4. Predicted surface rate of forward spread in areas of intermediate invasion by African Olive using two different models: (a) the BehavePlus model and (b) the Grassland Fire Danger Meter at 5, 10, 15 and 20% dead fuel moisture (DFM) conditions under increasing wind speed conditions.

Flame length

For the BehavePlus model, the maximum flame length predicted for 5% DFM and 50 km h^{-1} wind speed was 2.5 m and decreased with increasing DFM (Table 5.5). Consequently, maximum flame length predicted using the same wind speed scenario was 1.7, 1.3 and 1.2 m for 10, 15 and 20% DFM, respectively (Fig. 5.5a).

When using the GFDM model it was necessary to transform the data from flame height to flame length using the equations provided by Albini (1981). The maximum flame length predicted for 5% DFM and 50 km h^{-1} wind speed was 4.2 m (Table 5.5). The maximum flame length predicted using the same wind speed scenario was 3.7 m at 10% DFM, 3.3 m at 15% DFM and 2.8 m at 20% DFM (Fig. 5.5b).

Fireline intensity

Predictions of fireline intensity for areas of intermediate invasion were possible using the BehavePlus model. For 5% DFM and 50 km h^{-1}, the fire line intensity was 1887 kW m^{-1}. For 10% DFM, the intensity declined to 798 kW m^{-1}, 495 kW m^{-1} at 15% DFM and 386 kW m^{-1} at 20% DFM (Fig. 5.6).

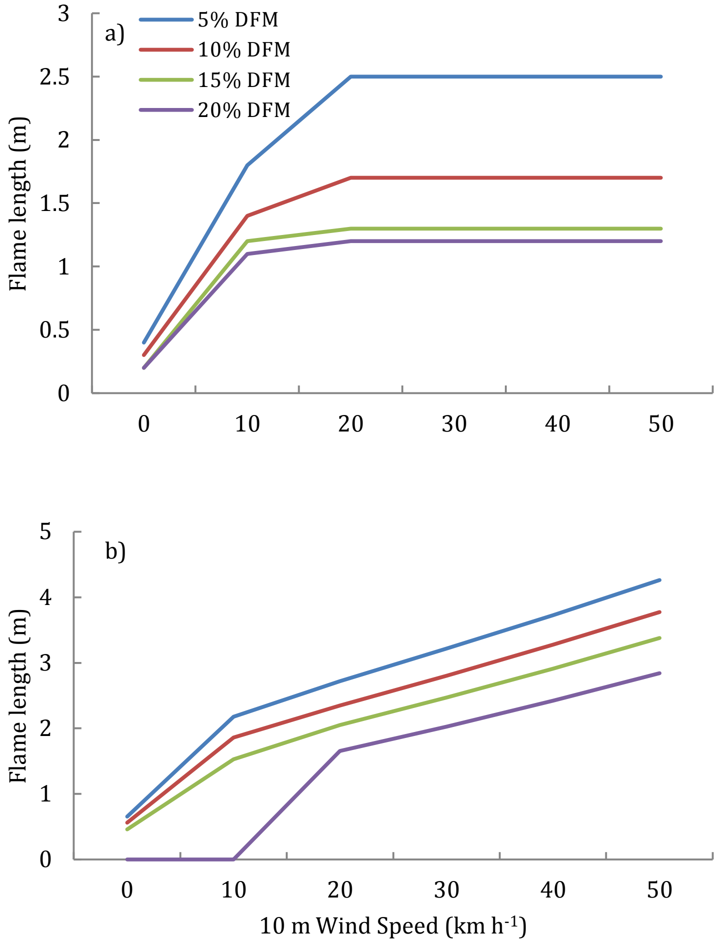

Figure 5. 5. Predicted flame length in areas of intermediate invasion by African Olive using two different models: (a) the BehavePlus model and (b) the Grassland Fire Danger Meter at 5, 10, 15 and 20% dead fuel moisture (DFM) conditions under increasing wind speed conditions.

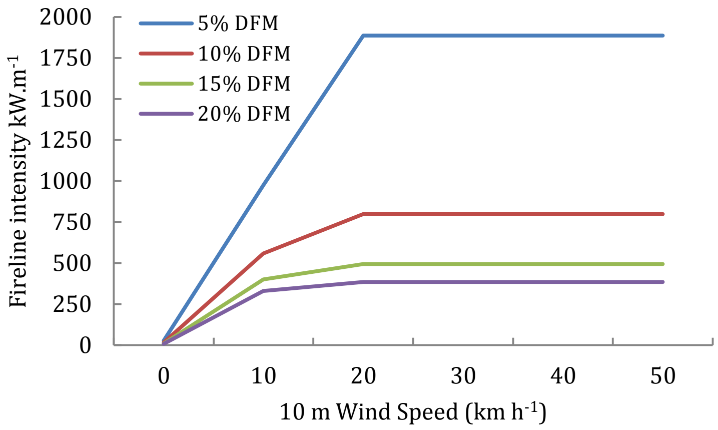

Figure 5. 6. Predicted fireline intensity in areas of intermediate invasion by African Olive using the BehavePlus model at 5, 10, 15 and 20% dead fuel moisture (DFM) conditions under increasing wind speed conditions.

5.3.3. Predicted fire behaviour in sites invaded with African Olive

Prediction of fire behaviour for areas with long-term invasion by African Olive (LI) were made using the BehavePlus model. The mean depth of the fuel bed in LI was 0.022 ± 0.006 m and was composed mostly of intact leaves of African Olive (see Section 2.3.3). Predicted fire behaviour in LI using the BehavePlus model varied strongly with changes in fuel bed depth. However, for the fuel bed depth measured in the field, regardless of the DFM scenario or surface wind speed used, the simulations did not show any fire spread (Fig. 5.7a, d, g).

When the fuel bed depth was changed to 0.06 m, the predicted fire behaviour was similar to that observed during the prescribed fire. The prescribed fire happened on 11 March 2013 at 12:25 pm with northerly wind conditions (10 km h^{-1} wind speed measured at the site), local temperature of 26 ˚C, and relative humidity of 44% showed very slow fire development. The average flame height was 0.23 ± 0.15 m with the surface layer acting as the main fuel source for the fire with occasional burning of clusters of near surface fuel. The ROS of the prescribed fire was 0.15 m min^{-1}. The model showed a similar ROS with the prescribed fire but there are no other published data relating to fire behaviour in pure stands of African Olive for comparison.

For simulations using a 0.06 m fuel bed depth, a wind speed of 50 km h^{-1} and 5% DFM scenario, the maximum surface rate of spread was 0.1 m min^{-1}, the flame length was 0.1 m and the fireline intensity was 1 kW m^{-1}. At 10% DFM, the predicted results were the same as for 5% DFM while no ROS was predicted for the 15% DFM scenario (Fig. 5.7b, e, h).

The height of the near surface fuel measured for LI was 0.19 ± 0.09 m (see Section 2.3.3). This fuel layer was mainly composed of seedlings of African Olive forming a non-continuous layer of live fuel. Although the fuel layer was not uniform, the height of this layer was used as the input for fuel bed depth in the model in order to explore the possible differences in predictions. In the case where the fuel bed height was 0.19 m, fire behaviour predictions were different for each scenario (Fig. 5.7c, f, i). The maximum surface ROS with a 50 km h^{-1} wind speed for 5% DFM was 2.2 m min^{-1}. At 10% DFM, this speed decreased to 1.6 m min^{-1} and at 15% DFM, the speed was 1.1 m min^{-1}. The predicted flame length was 1.5, 1.3 and 1.0 m for 5, 10 and 15% DFM, respectively. The fireline intensity under a 50 km h^{-1} wind speed was 268, 176 and 103 kW m^{-1} for 5, 10 and 15% DFM, respectively.

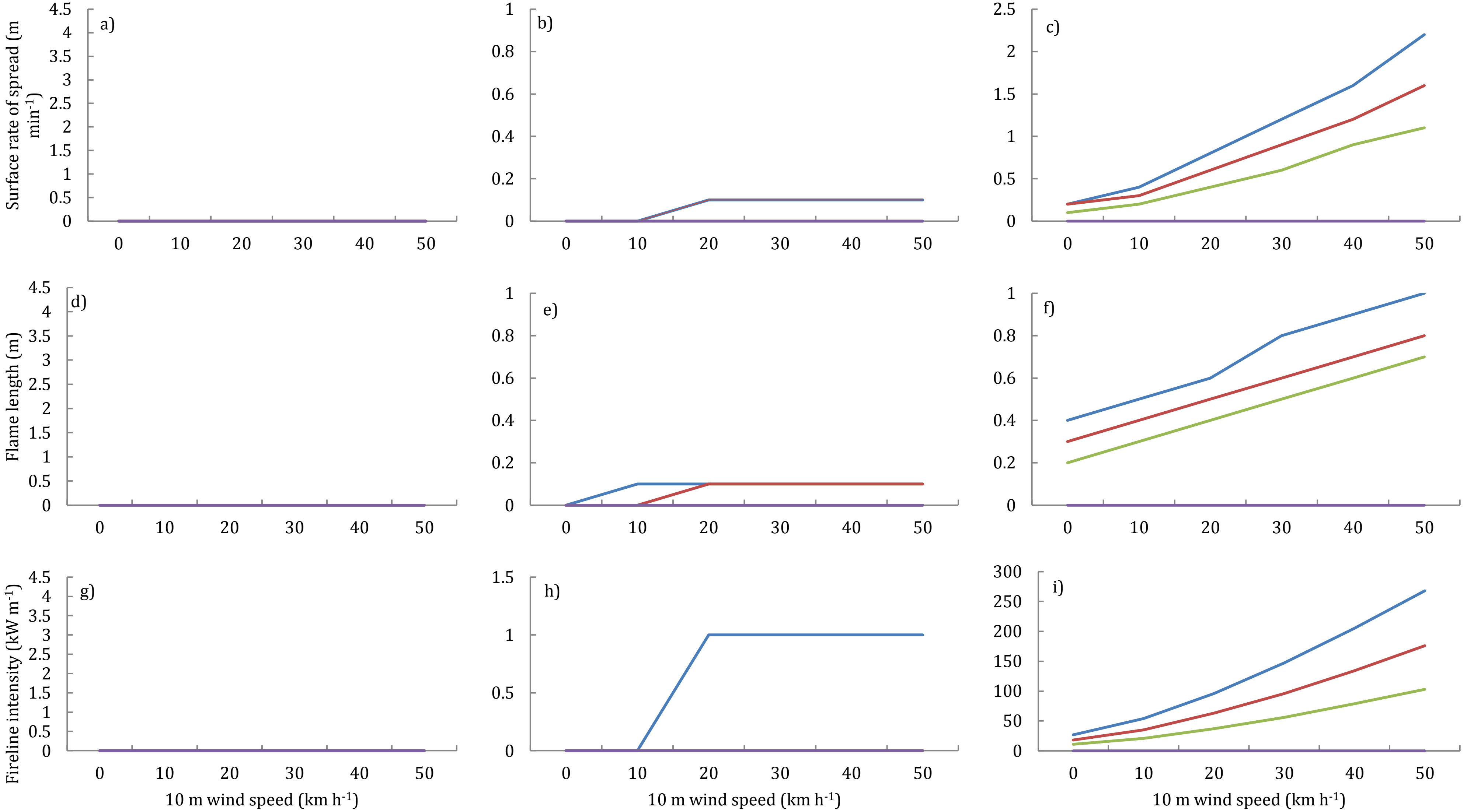

Figure 5. 7. Predictions of (a, b, c) surface rate of spread; (d, e, f) flame length; and (g, h, i) fireline intensity for long-term invasion by African Olive using the BehavePlus model for three different fuel depths: 0.02 m (a, d, g), 0.06 m (b, e, h) and 0.19 m (c, f, i), respectively.

5.4. Discussion

5.4.1. Inter-model comparisons: Cumberland Plain Woodland

Rate of spread

The differences in ROS predictions among the Vesta, FFDM and BehavePlus models for CPW varied according to fuel moisture. At the high end of the wind speed prediction scenario and 5% DFM, the Vesta model predicted a ROS that was 86% faster than the predictions made by the BehavePlus model and 88% faster than the FFDM model. At the highest DFM, the discrepancy in predictions between the Vesta and BehavePlus models diminished slightly. At 10% DFM, predictions of ROS were more than 70% faster than predictions made by the BehavePlus model and 92% faster than the FFDM model. At 15% DFM, the ROS predicted by the Vesta model was 57% and 94% faster than BehavePlus and FFDM models, respectively. At 20% DFM, the predicted ROS was 44% faster for the Vesta model compared to the BehavePlus model and 97% faster than the FFDM model.

Each model used in this study was built using a different framework and varied according to the inputs required to generate ROS predictions. The Vesta model used the fuel hazard score of the surface and near-surface layer to generate the ROS to predict fully developed large-scale fires (Gould *et al.* 2007a). Both FFDM and BehavePlus models used a description of the fuel to predict ROS. The Vesta model was developed using fully developed fires taking into account fuel structure while the FFDM model was developed using small fires in areas containing none or little understorey. In comparison, the BehavePlus model was developed using laboratory-based fires. Such differences in model development have implications for ROS predictions. Predictions of ROS using the BehavePlus model were derived from basic fire knowledge, fuel physics and combustion and thermodynamic principles supported by laboratory test fires

(Gould 1991). Rate of spread predicted by the Rothermel model and the associated predictions of flame length are highly influenced by the surface-area-to volume ratio and fuel height (Gould 1991). However, fuel load does not influence ROS but can strongly affect predictions of flame length. In comparison, the FFDM model (McArthur 1967) assumes that the ROS is strongly connected to the fuel load. Gould (1991) suggested that the overall algorithm relating ROS to windspeed used by the Rothermel model would cause the model to under-predict ROS at low wind speeds and over-predict at high wind speeds. Andrews *et al.* (2013) revised the wind limit function used in the Rothermel model and suggested that to avoid potential under-prediction of fire behaviour neither the original nor the improved revised wind limit should be imposed on the spread rate calculations and that the modelled ROS should not exceed the effective midflame wind speed.

The advantages and disadvantages of each model were compared for the different vegetation types investigated. The Vesta model was developed to predict fire spread in dry eucalypt forests with litter and shrub understorey but has not been validated in forests with a predominantly grassy understorey (Gould *et al.* 2007b), as is the case in many areas of the Cumberland Plain Woodland (Benson and Howell 2002; Watson *et al.* 2009). Although the model presents good quality results it assumes that predictions are for fires burning under summer conditions on fully developed fires (head fires of around 100 m) and when wind speed is less than 20 km h^{-1} (Gould *et al.* 2007b). The predictions made by the Vesta model in this study are therefore likely to be over-/underestimates given the assumptions needed by the model and the differences found in the weather conditions and in CPW vegetation. Gould *et al.* (2007a) showed that the Vesta model was validated to predict fire spread with accuracy of ±25% as long as the weather conditions are within the range measured by their work and the vegetation is a dry eucalypt forest. Cruz and Alexander (2013) analysing 1278 individual model predictions versus observations found an under-prediction in 64% of the cases. The main discrepancy found among the predictions made by the Vesta, BehavePlus and FFDM models is possibly being caused by the misprediction associated with the latter two models.

Cruz and Alexander (2013) analysed 49 studies comparing pairs of fire spread predictions and the related observations of fire behaviour. Thirty of the 49 studies used the Rothermel (1972) Fire Spread model to predict this characteristic. Only 16% of these studies predicted fire behaviour with accuracy of ±30%, half of the studies had an accuracy ranging from ±51–75%. Independent of the level of accuracy of the predictions using Rothermel model it is still important to test the results obtained in this work and compare them against real fires.

Each of the models used in the current study was developed with a specific aim and came from a different research background. Researchers developing the precursor of the BehavePlus model in the United States had engineering backgrounds and viewed fires as being a series of physics and fluid dynamic problems to be solved. In Australia, the researchers involved in studying fires were mostly from a forestry background. Moore (1986) pointed out that this basic difference was the major cause of most of the contrasting elements between the Behave and FFDM models leading to differences in flame length prediction. Although there are statistical differences among predictions, it is important to highlight that the FFDM model assumes that the fuel loading is 12.5 t ha^{-1}. The FFDM can be adjusted for alternative fuel loads by dividing the actual load by the fuel weight assumed in the development of the FFDM. Burrows (1994) showed that there is underprediction of ROS when wind speeds are low and fuel quantities are high for both models. Burrows (1994) claims that even though both models assume a direct relationship between fuel quantity and ROS, the FFDM model seriously underpredicts the ROS of fires burning at high wind speeds and low fuel moisture and only a marginally better prediction can be obtained using the Rothermel model, a result supported by the current study.

Flame length

The predicted flame length for the Vesta, FFDM and BehavePlus models for CPW varied according to fuel moisture. At the lowest DFM (5%), the flame length predicted by the Vesta model was 86% longer than the predictions made by the BehavePlus model and 26% longer than the predictions made by the FFDM model. At the highest DFM (20%), the prediction from the Vesta model was 75% longer than the BehavePlus model and 8% longer than predictions from the FFDM model.

The main reason for the large differences in predicted flame lengths is due to the way each model incorporates fuel arrangement. The flame length predicted in the Vesta model takes into consideration the height of elevated fuel and ROS. The surface module of BehavePlus model assumes the fuel is a horizontally uniform bed and calculates the flame length from fireline intensity. Even though fuel depth can be used to influence the flame length on BehavePlus this is particular exception on the way the fire behaviour is modeled in this software and would not be representative for this specific Australian vegetation due to fuel vertical discontinuity. Similarly, flame length predicted by the FFDM model assumes the fuel structure as a dry eucalypt forest without a developed elevated fuel layer (Moore 1986).

Each model used in this study has its own limitation for predicting flame length. Although the relationship between flame length and fire intensity is not linear it is still provides firefighters and fire managers with a powerful way of understanding the behaviour of a flame. Experienced personnel are needed when evaluating these variables and deciding which model should be applied in each type of vegetation. The latest empirical models, in this case the Vesta model, tended to present more realistic predictions for flame length in this vegetation type.

The BehavePlus model was originally developed for fuels in the United States and the system uses the Byram flame length-intensity relation to predict surface fire flame length (Alexander 1982). Cruz and Alexander (2010) showed that the Byram flame length-intensity relationship can produce results that range from underprediction to good approximations to reality for some fuel types but consistently underpredicts crown fires. The Byram flame length-intensity relationship was developed from a single field study in one fuel type. However there are at least another 19 flame length-intensity relationships described in the literature with a wide variety of outputs (Alexander and Cruz 2012) so the predictions presented here could be modified using a different relationship if it was found to perform better in a given fuel type.

The calculation used for flame height in the Vesta model gives a reasonably good prediction when the flame height of surface fires is up to 8 m (Gould *et al.* 2007b). However, when flame heights exceeds this limit there is the likely to be torching or crown fires in the intermediate and overstorey canopies depending on the bark hazard and density of the intermediate and overstorey layers. The trigonometric relationship between flame height and flame length always make the flame length longer than the measured flame height when flame is tilted. In this study, the prediction for the greatest flame length was given using the Vesta model and, at times exceeded 8 m. However, it is important to note that the vegetation types for which the predictions were made had no intermediate canopy layer. Therefore the chance of a crown fire developing is diminished due to the size of the gap between the surface fuel bed and the canopy and also to the very discontinuous canopy cover.

As shown for ROS, the FFDM model tends to underpredict flame height due to absence of a shrub fuel in the experimental fires used in its development (Burrows 1994; 1999a). Fire intensity and flame length are related by a power function (Burrows 1984) and the model coefficient used by the FFDM model is significantly different from those derived from the Byram flame length-intensity relation. The coefficient differences are likely to vary between different fuel types so flame length is not a reliable estimator of fire intensity when comparing fires in different fuels (Burrows 1984).

5.4.2. Inter-model comparisons: intermediate invasion

There were differences for ROS predictions between the BehavePlus and GFDM models for the II fuel type. At the high end of the wind speed prediction scenario and DFM of 5–15%, the predicted ROS from the GFDM model was 51–57% faster than from the BehavePlus model. The predicted ROS for the GFDM model was similar or slower than the predictions from the BehavePlus model only when DFM was 20% or higher. From 5–15% DFM, the flame lengths predicted by the GFDM model were approximately two times longer than for the BehavePlus model. The differences in ROS predictions are mainly explained by the influence of fuel bed depth and bulk density of the fuel. For the BehavePlus model this characteristic can be a direct input from field measurements or derived from one of the fuel type models described by Scott and Burgan (2005).

The ROS predicted by the GFDM model is based on work from McArthur (1966) and Noble *et al.* (1980) with improvements by Cheney *et al.* (1998). Work done by

Cheney *et al.* (1998) considered not only the packing ratio of fuel but also incorporated adjustments for the degree of curing, pasture condition and the influence of wind. Rate of spread predictions using the BehavePlus model use the relationships developed by Rothermel (1972) and the fuel models of grass-shrub vegetation type developed by Scott and Burgan (2005) are closest to II areas.

Areas of II were covered by grasses with young individuals of African Olive scattered throughout (see Chapter 2). In this vegetation type, the grass fuel is the main driver of the fire front. Fires in continuous natural grassland tend to have a fast ROS, develop very quickly and react to wind speed and direction almost instantly (Cheney and Sullivan 2008). Both models predicted ROS within the normal range of speed recorded for grassfires in Australia (Noble 1991; Cheney and Sullivan 2008).

Grassfire predictions are relatively simple when compared to fires in forest, woodland or shrubland. The architecture of grass makes fires in this vegetation type highly responsive to changes in the weather which give firefighters the feeling that the fire behaviour is erratic and hard to forecast (Cheney and Sullivan 2008). Despite this, if weather variables are known or can be accurately measured, the fire behaviour of grassfires can be predicted reasonably well (Cheney *et al.* 1998; Cheney and Sullivan 2008).

Flame length predictions made using the GFDM model were double that made using the BehavePlus model for all moisture scenarios. The GFDM model does not predict fireline intensity however the values predicted by the BehavePlus model were within the range reported for Australian savannas (Griffin and Friedel 1984). Flame length and fire intensity are important variables for fire managers and firefighters to know and it is possible to calculate the size of fire breaks according to predicted flame intensity and length (Wilson 1988).

The effects of young individuals of African Olive in areas of II were not captured by the models due to the way the fuel is taken in account, however, it is unlikely that the fire behaviour would be altered due to their presence. Grassland areas invaded with African Olive are more likely to burn in a patchy pattern leaving islands of non-burned areas around the young trees due to their high moisture content (personal observation). The fast nature of grassfires would not allow enough time for the majority of the tree to heat up to lethal temperature. However, von Richter (2005) showed that low intensity fires with flames height up to 1.2 m killed 100% of Olive with stem diameter smaller than 5 mm and 80% of Olive measuring less than 20 mm diameter suggesting that fire could be used as a means of control in initial stages of invasion.

Even though it is possible to achieve good predictions in this vegetation type with a custom made fuel type using the BehavePlus model, the GFDM model is still widely used by fire managers in Australia. This study has shown that experienced professionals could use results from both models to achieve better results during prescribed fire in areas invaded by woody weeds.

5.4.3. Long-term invasion by African Olive

Due to the lack of information for fire behaviour in stands of African Olive one of the aims of this study was to build a fuel model able to represent this vegetation type and to compare the possible outcomes of fire prediction using different weather scenarios. The BehavePlus model was used to predict fire behaviour in this new fuel type. However, due to the large variability in the litter layer depth in LI and the sensitivity of the BehavePlus model to this variable, three vegetation models were built varying the fuel bed depth in each of them allowing comparison of how each depth would affect fire behaviour predictions.

The first fuel model used an average fuel bed depth of 0.022 m (the measured fuel bed depth) and did not show any fire spread (0 m min^{-1} ROS and 0 m flame height). In the second fuel model, a depth of 0.060 m was used as this is the minimum fuel depth used for models created by Scott and Burgan (2005). This model predicted values close to what was measured during a low intensity prescribed fire. The BehavePlus model uses fuel depth as an input value to determine bulk density of the fuel bed which is an intermediate value to Rothermel's Surface Fire Spread model (White *et al.* 2013a). In this way, a small change in the fuel depth can cause a major change in the bulk density of the fuel and lead to a misprediction of how a fire would behave in the vegetation. In the third fuel model, the average height of understorey seedlings present was used as the fuel bed depth. This model predicted the fastest and longest flames with reasonable results for fire intensity. Seedlings of African Olive form a dense matt under parent trees (Cuneo and Leishman 2006) and could possibly burn during a prescribed fire due to their smaller diameter (von Richter *et al.* 2005). However, even though this "layer" of thin live twigs could catch fire, it is not continuous and is not likely to carry fire for any distance making this an unrealistic fuel scenario.

Establishing scenarios grounded in realistic field values and measurable parameters made it possible to estimate fuel load that, when associated with topography and weather, allowed rational predictions of fire behaviour (White *et al.* 2013a). As there are no Australian models or methods that can accurately measure and describe fire behaviour in a novel fuel type, the use of software that allows the construction of fuel models (e.g. the BehavePlus model) was warranted.

5.5.2. Management implications

This is the first study to compares fire behaviour predictions from BehavePlus, FFDM, Vesta and GFDM models in Australian fuels. There have been previous attempts to compare the FFDM and Behave models (Moore 1986; Gould 1991; Burrows (1994) but there are no published studies as this one. Understanding the drivers of flammability and the best way accurately to predict fire behaviour is becoming more important. New plant species are still being transported across continents and have the potential of becoming invasive (Groves 2006). With this the risk of fuel structure and load and flammability alteration poses a new challenge to fire scientists as making fire behaviour predictions is extremely difficult.

There are intrinsic limitations when it comes to fire modelling independently of using empirical or physical models (Sullivan 2009a, 2009b). This study shows that, in some instances, fire behaviour predictions made by the BehavePlus model were close to what was predicted using the Vesta and FFDM models. For example, the BehavePlus and FFDM models had similar ROS at higher fuel moisture (20%). This indicates that the manipulation of this factor could be used in both models to fine tune the fire behaviour prediction process to improve the results (Matthews 2010; Matthews *et al.* 2010). Similarly, the predictions made with the GFDM and BehavePlus models showed some similarities and can be used together to improve management of areas that have grasses as the main driver for fire spread.

Recent studies have concluded that the BehavePlus model often underestimates fire behaviour (Streeks *et al.* 2005; Stephens *et al.* 2008; Cruz and Alexander 2013; White *et al.* 2013a). Although this model has limitations, it is important to highlight that its efficiency can be improved when the values of the fuel characteristics are directly measured and used as inputs for the fuel bed models (i.e. this work) instead of using standard fuel models (Grabner *et al.* 1997; White *et al.* 2013a). When used by experienced fire managers, the BehavePlus model proved to be a strong tool with a friendly user interface that is able to produce insights about the potential fire behaviour for novel fuel types (White *et al.* 2013a).

The fire behaviour predictions for native fuels made by the BehavePlus model in this study tended to underestimate the results when compared with empirical approaches of prediction such as the Vesta and FFDM models. The underestimation of the fire spread is mostly due to the difficulty in quantifying the fraction and packing ratio of the litter composing the fuel layer which is the main driven for fire spread (Cruz and Fernandes 2008). The BehavePlus model was developed to predict fires in fuels composed of leaves of softwood from the northern United States. These leaves are considerably different from the broad, sclerophyllous leaves that compose most of the litter in Australian forests. Although the BehavePlus model allows the customisation of fuel models (Andrews 2013), the structure of Australian fuels and limited litter depth, rarely exceeding 5 cm, tend to increase the fuel consumption producing longer flame lengths and higher fire intensities (Moore 1986).

When using the BehavePlus model, determining the fuel bed depth is uncertain. Even though this variable was measured in the field, the terrain and slope can create substantial variations during the burning process accelerating or slowing flames through the fuel over time. Differences in fuel moisture among vegetation types for given weather conditions was also not considered and more studies involving this aspect of the fuel are still needed. To fine tune the construction of fuel models to improve accuracy of predictions more observations of prescribed burns under different weather conditions are needed. The administration and funding needed to establish large-scale burn experiments to validate fire spread models is extremely difficult and costly due to the risk associated with projects such as this.

Rothermel (1991) found that flame length is not a set parameter and can vary according to the observer measuring it. This uncertainty would make it a poor predictor of fire behaviour from a scientific and engineering point of view. However, as this variable is readily observed on the fire ground and it provides a visual cue for fire intensity, it is worth including it as a primary fire variable.

The species *Olea europaea*, of which African Olive is a subspecies, originated in Mediterranean regions of southern Europe and northern Africa. Wildfires in Mediterranean areas are known for their high intensity resulting in severe crown fires due to the high quantities of volatiles compounds present in leaves making them highly ignitable and combustible (Dimitrakopoulos 2001; Kozlowski 2012). The history of the species *O. europaea* and its domestication, cultivation and diffusion across the Mediterranean region is extremely complex (Terral *et al.* 2004). Olives have had millions of years to co-evolve with other species in their natural distribution (Lumaret and Ouazzani 2001). In its natural environment growing alongside other species, *O. europaea* can develop conditions for vertical continuity of the fuel which is fundamental for carrying fire to the crown.

From the work described here, the possibility of a crown fire occurring in a stand of African Olive is relatively low unless extreme conditions take place. However, there are reports of severe fires happening in stands of African Olive in Australia (personal communication, G. Douglas, September 2013). Management authorities in highly infested areas such as in the Adelaide Hills in South Australia and Mount Annan in New South Wales are concerned about the damage that potential fires happening in these areas could cause to human assets and native vegetation in the surroundings areas

(Government of South Australia 2001; Olives 2004). In Australia, *O. europea* acts as an environmental engineer diminishing the canopy cover of natives by 80%, suppressing regeneration and eliminating native shrubs and ground cover species by 50% (Crossman 2002; Cuneo and Leishman 2012). The lack of fine fuel in the near surface layer and low amount of readily available fuel is likely to be the main reason for differences in fire behaviour in Mediterranean regions where *Olea* sp. is native and in Australia where it has become a highly invasive woody weed.

6. General conclusions and management implications

6.1. General conclusions

The Australian flora is estimated to have around 25 000 plant species and the number of introduced plant species is thought to be at least equal to this (Groves 2002). African Olive (*Olea europaea* ssp. *cuspidata*) is recognised as a noxious weed in NSW and invasion by it is listed as key threatening process as these plants pose a potentially serious threat to threatened species. Besides, this species can negatively influence primary production, the environment or human health and must be fully suppressed and destroyed (*Noxious Weeds Act* 1993). Apart from the profound ecological impact caused by African Olive (Cuneo and Leishman 2012), no previous work has investigated how these plants can also alter the fuel structure and fire behaviour of invaded areas. This study has shown that vertical changes in fuel distribution promoted by African Olive associated with a deeper litter layer and a different light environment under the trees can have an important impact on fire behaviour in invaded areas. These characteristics are in addition to the increased sustainability of the combustion process in this fuel type associated with a greater effective heat of combustion than that measured for native Australian vegetation types. The combination of structural alteration of the fuel and longer flaming periods (sustainability) suggests that, in case of a fire event in this fuel type, the fire severity and ecological effects caused by it can be largely damaging to the environment and neighbouring assets. Even though fire behaviour models show much slower and less intense fires for areas invaded by African Olive compared to areas of intermediate invasion and Cumberland Plain Woodland, in a worst-case scenario, this vegetation could sustain a fire for long periods potentially causing a chain of damaging effects to the ecosystem and surrounding areas.

Alteration of the vertical fuel distribution was also found in areas invaded with Cootamundra Wattle (*Acacia baileyana*). This species is considered an environmental weed under the *Pest Plants and Animals Act* 2005 in the Australian Capital Territory (ACT). The growth of the plant must be managed in a manner that reduces its numbers, spread and incidence and continuously inhibits its reproduction. In addition, the plant may not be sold, propagated or knowingly distributed. The presence of Cootamundra Wattle in Yellow Box-Red Gum Grassy Woodland in the ACT causes the vertical structure of the fuel to shift upwards towards the canopy and to be more evenly distributed vertically. This promotes a higher fire hazard in invaded sites compared to natural uninvaded areas. Although this woody weed causes alterations to the fuel structure and increases the fire hazard, the results presented indicate that flammability of this species does not differ from the native vegetation. Any changes in fire behaviour in these areas are likely to be due to an altered fuel structure and denser arrangement of fuel instead of changes in the effective heat of combustion or any other flammability-related components.

The differences in fuel load, structure and flammability found between African Olive and Cootamundra Wattle suggests that flammability traits could possibly relate to plant origin and further investigation of this factor would worth include in future studies. Studies investigating correlations between leaf morphology and flammability are becoming more important as authorities have to deal with the consequences of plant invasion and the awareness of the general public increases. The characterisation of leaf traits from a range of species reinforced the notion that there are intrinsic differences among invasive and native plants. However we did not find strong correlations between leaf size and flammability traits as suggested in other studies. It seems more appropriate to consider the flammability of leaves as being independent from or only having weak relationships with leaf morphology. The research presented shows that, in order to understand plant flammability, it is necessary to measure all of the components of flammability and a range of related variables. This type of analysis can be used to rank plant flammability to compare potential effects of burning woody weeds with native vegetation. A number of caveats such as plant structure still needed to be included in such a ranking system but as a first effort, this approach can conceivably be used in future fire management strategies.

For woody weeds, the characterisation of fuel architecture and fuel load combined with a strong data set describing flammability could provide robust inputs to feed existing models to predict fire behaviour in novel vegetation types. Testing and adapting existent fuel description methodologies used to describe Australian fuels (e.g. the Vesta method) in novel fuel types is also fundamental for comparing possible prediction outputs amongst different fire behaviour models and what is found in prescribed fires or wildfires.

The science involved in prediction of fire behaviour in Australia has mostly been developed through empirical models designed for and tested in native fuels (Cheney *et al.* 1993, Gould *et al.* 2007a, Cruz *et al.* 2010, Gould *et al.* 2013). Accurate fire prediction in areas invaded by woody weeds requires new fire spread models. Using new knowledge of leaf flammability and appropriate fuel description variables to build custom fuel type models and simulation of fires in quasi-physical models such as the BehavePlus model can provide fire managers with a powerful tool to reduce risk and to better apply their resources.

6.2. Fire behaviour prediction limitations

The description of fuel loads is one of the most important variables used to develop fire behaviour models (Keane 2012). Fuel description systems try to catalogue information about the fuel bed in a logical way in order to use these variables as inputs to software or models capable of simulating/predicting fire behaviour and danger, fire effects and smoke emissions (Deeming *et al.* 1977; Anderson 1982; Sandberg *et al.* 2001; Gould *et al.* 2007a; Arroyo *et al.* 2008).

Keane (2012) summarises the approaches, methods and systems used to describe wildland surface fuel loading and how fire behaviour prediction models take them into account. This review summarises the limitations involved in actual fuel description systems and calls attention to the fact that none of the current fuel systems can be used in all phases of fire management, such as predicting fire spread and danger, estimating emissions and calculating flame length and intensity. There is a growing need for new methodologies that categorise and describe fuels in a simplified and cheap way but are also capable of detailing the complexity of the systems being observed (Keane 2012, Wise and Wright 2014).

For a long time Australian fire authorities have used the models developed by McArthur (1962, 1967) which was developed using small localised fires to predict fire behaviour in forested vegetation. The Vesta model (Gould *et al.* 2007a) brought into perspective a new way of describing Australian fuels by dividing the fuel in different strata and ranking it according to a hazard score. This methodology is relatively easy and cheap, however, there are limitations in terms of the skill of the person making the assessment and in differences in perception between two different surveyors (Watson *et al.* 2012). Inconsistency in scoring fuel hazard can possibly lead to discrepancies in an array of management applications and can affect fire fighting safety and effectiveness (Watson *et al.* 2012).

Using the BehavePlus model to build a custom fuel type and simulate fires to predict fires in invaded areas seems to be an effective way to gain an insight to fire behaviour in novel vegetation types. For example, Dimitrakopoulos (2002) used the BehavePlus model to build customised fuel models for 181 distinct vegetation types and to simulate fire behaviour in these areas. However, the BehavePlus model has many limitations including the cost and effort needed to sample the vegetation in order to properly calculate the inputs and the way in which the mechanics of the model work. Although the BehavePlus model was developed using laboratory fires it permits entry of basic fuel model parameters, allowing analysis of how changes in various fuel variables (e.g. fuel bed depth, fine fuel SA/V, live fuel load, heat content) affect modelled fire behaviour (Andrews 2013). Due to many ecophysiological and structural singularities, each species of invasive plant will alter fuel loads of their surrounding environment in a very particular way. Having a tool like the BehavePlus model to simulate fire behaviour of invaded areas represents an important step forward in terms of management. However, validation of the outputs with measurements from real fire is still needed to adjust the models.

6.3. The National Weeds Strategy, prescribed burning codes and plant flammability

Invasive plant species or weeds are considered to be one of the biggest threats to communities and ecosystems causing a considerable biodiversity loss across the globe (Vitousek *et al.* 1997; Pyšek *et al.* 2009; Gaertner *et al.* 2014). The presence of invasive species causes impacts at all levels of biological organisation (Vilà *et al.* 2011), decreasing the local biodiversity (Gaertner *et al.* 2009; Richardson and Gaertner 2013), altering the productivity of ecosystems (Richardson and Gaertner 2013), and changing nutrient cycling (Liao *et al.* 2008) and fire regimes (D'Antonio and Vitousek 1992; Brooks *et al.* 2004; Mandle *et al.* 2011).

In Australia, weeds have major impacts on the economy, environment and society by damaging natural landscapes, agricultural lands, water ways and coastal areas (Australian Weeds Strategy 2007). Estimates of the cost of weeds to the agricultural sector total around AUD$4 billion per year and while there are no estimates of the cost of weeds on conservation and biodiversity, it is believed that the value is around the same amount (Australian Weeds Strategy 2007).

The National Weeds Strategy Executive Committee (hereafter referred to as 'the Committee') was established in 1997 with the objective of providing a framework to reduce the economic, environmental and social impact of weeds (Australian Weeds Committee 2005). In the same year, the National Weeds Strategy (hereafter referred to as 'the Strategy') was released. The Strategy aimed to reduce the impact of weeds and to strengthen the cost efficiency and effectiveness of weed management. To implement the Strategy, the Committee identified that the biggest impact from weed problems in Australia related to the effect and spread of individual species (Thorp and Lynch 2000). Based on this conclusion, the work to develop a list of Weeds of National Significance (WONS) was started. In 1999, the inaugural list of WONS was announced and contained the 20 most important species at a national level. Since then, the Strategy has been replaced with a revised version called the Australian Weeds Strategy, and in 2012, an additional 12 species were added to the list of WONS. For each weed species on the list, there is a management guide and a national Best Practice Management manual containing basic information of the impacts and the best ways to manage/eliminate the species.

At a state level, the legislation dealing with weeds and the ways in which each state classifies its weeds of importance varies. The WONS still has an overall priority, however each state can have its own legislation to deal with weed species with high local impact that are not necesserily listed as a WONS.

In general, the criteria used to classify weed species into different classes of importance do not take in account the impacts of these plants on the fire regime and fire behaviour. At present there is very little information on how invasive species can affect fire behaviour in Australia especially when it comes to woody weed species (Rossiter *et al.* 2003; Berry *et al.* 2011; Murray *et al.* 2013). Including information about plant flammability, describing the possible alteration in fuel structure and load and the fire-related life history for each invasive plant species into a "fire section" in the WONS and other weed classification systems (e.g. NSW Invasive Species Plan (Department of Primary Industries 2008)) could lead to a more comprehensive and effective aproach in the management of invaded areas and proper evaluation of weed impacts.

Studies describing the mechanisms by which weeds can alter the fire frequency are becoming more frequent (D'Antonio and Vitousek 1992; Brooks *et al.* 2004; Mandle *et al.* 2011), however it is hard to predict their effect on fire behaviour in infested areas. Generally, the invasion pattern of weeds start from roads or backyards into nearby vegetation (Higgins and Richardson 1996; Groves *et al.* 2005; Arteaga *et al.* 2009). The presence of large numbers of invasive species in patches of forest close to human assets could result in a fire that behaves in a completely different way from that expected for native forest leading to misscalculated fire suppression actions. If information relating to fire behaviour and fuel structure of woody weeds was readily available to fire managers, the safety of neighbouhoods near fuel hazard reduction zones or areas that are likely to suffer from wildfires would be enhanced.

Prescribed burns in NSW must be in agreement with the Bush Fire Environmental Code (NSW Rural Fire Service 2006) however, there is little material in the Code regarding fire management of weeds. Hazard reduction burns also need to be conducted in accordance with the NSW Rural Fire Service *Standards for Low Intensity Bush Fire Hazard Reduction Burning* (NSW Rural Fire Service 2003). The main problem encountered here is that guidelines relating to weeds in the RFS Standards do not consider that these plants can change fire intensity and severity. The RFS Standards states that prescribed fires in areas containing weeds should operate in accordance with the *Noxious Weeds Act* 1993 (New South Wales Government 1993), and the conditions in the Best Practice Management manual for the species involved must be imposed to prevent their spread. If information regarding plant flammability and alteration in fire behaviour is added to Best Practice Management manual for each species it could be used as a guideline for hazard reduction, thus improving the control of weeds.

Mandle *et al.* (2011) showed that intrinsic fuel properties of plants affect fire frequency, seasonality and intensity. According to their research there are two possible ways in which a woody weed can alter the fire regime. On one hand, woody weeds can cast shade, alter surface fuel load and composition and increasing its moisture, modify vertical fuel distribution and biomass and change the local microclimate. In some instances, woody weeds can reduce fire spread and diminish the risk of fires. However, an intense fire can still happen in such areas under particular conditions such as during extreme fire weather. On the other hand, fast growing plants forming dense stands of flammable material can promote and enhance the pre-existing fire regime possibly creating a feedback cycle exacerbating further invasion.

Woodlands in NSW invaded by African Olive seems to fit in the first category described by Mandle *et al.* (2011), while the invasion of woodlands by Cootamundra Wattle is more likely to fit into the second category. Both invasion scenarios have the potential to influence fire behaviour in very different ways. Even though describing and measuring the fuel provides the best information about the vegetation structure, there are no empirical models capable of predicting fire behaviour in invaded areas. Testing new methodologies to describe fuels and acquiring information on how each important weed in Australia could alter fuel loads, structure and flammability is becoming more important as the rates of plant invasions and spread rises (Groves 2002) and the global climate changes (Stocker *et al.* 2013). Consolidating this information into one database or adding it to nationally recognised databases and making the current knowledge available in established frameworks such as the Australian Weeds Strategy and Codes and Standards for prescribed burning would provide fire and land managers with a poweful tool for predicting fire behaviour and for integration of various aspects of fire fighting and weeds management industries.

6.4. Recommendations and further research

Investigate the alteration of fuel by weeds

To improve our understanding of the processes by which weeds can alter the fuel composition and structure more studies are needed to describe the fuel in invaded areas. Data about fuel load and vertical and horizontal structure of fine fuels in invaded areas needs to be systematically added to a common database allowing better forecasting of fire behaviour in these areas. Starting with the WONS and including this information into relevant Best Practice Management manuals would improve the knowledge and management of invaded sites.

Improved understanding of flammability of native and introduced species

Data collected in this study suggested that there is a wide range of flammability-related variables capable of influencing fire behaviour. New studies describing plant flammability should include accurate measurements of leaf morphology and quantification of the four components of flammability. This additional data will enable further analyses of the correlation between leaf morphology and flammability. Investigating more woody weed species using a standardised methodology capable of accurately measuring the four components of flammability could support the creation of a ranked list of species flammability. White and Zipperer (2010) deeply discuss the usage and worthlesness of creating a plant list like suggested in this work. Despite the polemics on the scientific and policy making values of creating plant lists it is important to highlight that there is an increasing demand for them as more people move in the interface beetween forests and urban lands in Australia. Lists like this could and should be used together with fuel structure data to support managers taking decisions on wether or not to burn an area.

Predictions and validation

Using fuel description data aquired in the field to build customised fuel models and simulation of fires in these fuels using the BehavePlus model is a reasonable option to aquire insight of the fire behaviour in weed invaded areas. However, caution is needed when using these models to predict fire behaviour and only experienced fire managers should use this as an operational tool. Validation of the models using fire behaviour data aquired from the field is essential to improve and adjust the results.

7. References

Aber, JD, Melillo, JM, McClaugherty, CA (1990) Predicting long-term patterns of mass loss, nitrogen dynamics, and soil organic matter formation from initial fine litter chemistry in temperate forest ecosystems. *Canadian Journal of Botany* **68**, 2201-2208.

Adair, RJ, Groves, RH (1998) Impact of enviromental weeds on biodiversity a review and development of methodology. National Weed Program: Camberra. 55.

Addo-Fordjour, P, Obeng, S, Addo, MG, Akyeampong, S (2009) Effects of human disturbances and plant invasion on liana community structure and relationship with trees in the Tinte Bepo forest reserve, Ghana. *Forest Ecology and Management* **258**, 728-734.

Aerts, R (1996) Nutrient resorption from senescing leaves of perennials: are there general patterns? *Journal of Ecology* 597-608.

Agee, JK (1997) The severe weather wildfire - too hot to handle? *Northwest Science* **71**, 153-156.

Albini, FA (1976) Estimating wildfire behavior and effects. Deptament. of Agriculture, Forest Service, Intermountain Forest and Range Experiment Station.

Albini, FA (1981) A model for the wind-blown flame from a line fire. *Combustion and Flame* **43**, 155-174.

Alexander, ME (1982) Calculating and interpreting forest fire intensities. *Canadian Journal of Botany* **60**, 349-357.

Alexander, ME, Cruz, MG (2012) Interdependencies between flame length and fireline intensity in predicting crown fire initiation and crown scorch height. *International Journal of Wildland Fire* **21**, 95-113.

Alexander, ME, Stocks, Bl, Lawson, BD (1996) The Canadian Forest Fire Danger Rating System. In: Initial Attack - the magazine for wildfire managment. Bombardier Inc., Canadair, Montreal. Spring. 6-9.

Allen, EB, Steers, RJ, Dickens, SJ (2011) Impacts of fire and invasive species on desert soil ecology. *Rangeland Ecology and Management* **64**, 450-462.

Allen, HD (2008) Fire: plant functional types and patch mosaic burning in fire-prone ecosystems. *Progress in Physical Geography* **32**, 421-437.

Anderson, HE (1970) Forest fuel ignitability. *Fire Technology* **6**, 312-319.

Anderson, HE (1982) Aids to determining fuel models for estimating fire behavior. *The Bark Beetles, Fuels, and Fire Bibliography* **143**.

Andrews, PL (1986) Fire behavior prediction and fuel modeling system - BURN subsystem, part 1. USDA Forest Service.

Andrews, PL (2009) BehavePlus fire modeling system, version 5.0. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Andrews, PL (2010) Do you BEHAVE? - Application of the BehavePlus fire modeling system. In: 3rd Fire Behavior and Fuels conference. Spokane, Washington.

Andrews, PL (2013) Current status and future needs of the BehavePlus Fire Modeling System. *International Journal of Wildland Fire* **23**, 21-33.

Andrews, PL, Bevins, CD, Seli, RC (2003) BehavePlus Fire Modeling System: Version 2.0: User's Guide. US Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Andrews, PL, Cruz, MG, Rothermel, RC (2013) Examination of the wind speed limit function in the Rothermel surface fire spread model. *International Journal of Wildland Fire* **22**, 959-969.

Anon. (2006a) National Vegetation Information System. Australian Government, Department of the Environment and Water Resources, Canberra. Retrieved 12 February 2015, from http://www.environment.gov.au/land/native-vegetation/national-vegetation-information-system.

Arroyo, J, Maranon, T (1990) Community ecology and distributional spectra of Mediterranean shrublands and heathlands in southern Spain. *Journal of Biogeography* **17**, 163-176.

Arroyo, LA, Pascual, C, Manzanera, JA (2008) Fire models and methods to map fuel types: the role of remote sensing. *Forest Ecology and Management* **256**, 1239-1252.

Arteaga, M, Delgado, J, Otto, R, Fernández-Palacios, J, Arévalo, J (2009) How do alien plants distribute along roads on oceanic islands? A case study in Tenerife, Canary Islands. *Biological Invasions* **11**, 1071-1086.

Asensio, MI, Ferragut, L (2002) On a wildland fire model with radiation. *International Journal for Numerical Methods in Engineering* **54**, 137-157.

Ashton, IW, Hyatt, LA, Howe, KM, Gurevitch, J, Lerdau, MT (2005) Invasive species accelerate decomposition and litter nitrogen loss in a mixed deciduous forest. *Ecological Applications* **15**, 1263-1272.

Atreya, A (1998) Ignition of fires. *Philosophical Transactions of the Royal Society of London Series* 2787-2814.

Australian Bureau of Agricultural and Resource Economic Sciences (ABARE) (2011) Australia's forests at a glance. Australian Government, Canberra. p 102.

Australian Capitol Territory Government (2014) Camberra Nature Park. Retrieved 01 July 2014, from http://www.tams.act.gov.au/parksrecreation/parks_and_reserves/canberra_nature_park

Australian Government Department of the Environment and Water Resources (2006) Australian Weed Strategy – a national strategy for weed management in Australia. Camberra ACT 24 p.

Australian Weeds Committee (2005) The National Weeds Strategy: a strategic approach to weed problems of national significance. Australian Government, 35 p.

Australian Weeds Committee (2014) Weeds Australia – An Australian Weeds Committee National Initiative. Retrieved 25 August, 2014, from http://www.weeds.org.au/.

Australian Weeds Strategy (2007) A national strategy for weed management in Australia. Natural Resource Management Ministerial Council, Commonwealth of Australia.

Babrauskas, V, Peacock, RD (1992) Heat release rate: the single most important variable in fire hazard. *Fire Safety Journal* **18**, 255-272.

Baker, TR, Chao, KJ (2009) Manual for coarse woody debris measurement in RAINFOR plots.

Balch, JK, Bradley, BA, D'Antonio, CM, Gómez-Dans, J (2013) Introduced annual grass increases regional fire activity across the arid western USA (1980–2009). *Global Change Biology* **19**, 173-183.

Bale, CL, Charley, JL (1994) The impact of aspect on forest floor characteristics in some eastern Australian sites. *Forest Ecology and Management* **67**, 305-317.

Baruch, Z, Goldstein, G (1999) Leaf construction cost, nutrient concentration, and net CO2 assimilation of native and invasive species in Hawaii. *Oecologia* **121**, 183-192.

Bate, LJ, Torgersen, TR, Wisdom, MJ, Garton, EO (2004) Performance of sampling methods to estimate log characteristics for wildlife. *Forest Ecology and Management* **199**, 83-102.

Behm, AL, Duryea, ML, Long, AJ, Zipperer, WC (2004) Flammability of native understory species in pine flatwood and hardwood hammock ecosystems and implications for the wildland–urban interface. *International Journal of Wildland Fire* **13**, 355-365.

Benson, DH (1992) The natural vegetation of the Penrith 1: 100 000 map sheet. *Cunninghamia* **2**, 541-596.

Benson, DH, Howell, J (2002) Cumberland Plain Woodland ecology then and now: interpretations and implications from the work of Robert Brown and others. *Cunninghamia* **7**, 631-650.

Benson, DH, Howell, J, Gardens, RB (1990) Taken for granted: the bushland of Sydney and its suburbs. Kangaroo Press. 160.

Berendse, F, Berg, B, Bosatta, E (1987) The effect of lignin and nitrogen on the decomposition of litter in nutrient-poor ecosystems: a theoretical approach. *Canadian Journal of Botany* **65**, 1116-1120.

Berg, B, De Santo, AV, Rutigliano, FA, Fierro, A, Ekbohm, G (2003) Limit values for plant litter decomposing in two contrasting soils—influence of litter elemental composition. *Acta Oecologica* **24**, 295-302.

Berg, B, McClaugherty, C (2003) Plant litter –Decomposition, humus formation, carbon sequestration. Berlin, DE. Springer. 338.

Berger, TW, Glatzel, G (1996) Deposition of atmospheric constituents and its impact on nutrient budgets of oak forests (*Quercus petraea* and *Quercus robur*) in Lower Austria. *Biological Conservation* **75**, 309-309.

Berry ZC, Wevill K, Curran TJ (2011) The invasive weed Lantana camara increases fire risk in dry rainforest by altering fuel beds. *Weed Research* **51**, 525-533.

Besnard, G, Henry, P, Wille, L, Cooke, D, Chapuis, E (2007) On the origin of the invasive olives (*Olea europaea* L., Oleaceae). *Heredity* **99**, 608-619.

Bird, RB, Bird, DW, Codding, BF, Parker, CH, Jones, JH (2008) The "fire stick farming" hypothesis: Australian Aboriginal foraging strategies, biodiversity, and anthropogenic fire mosaics. *Proceedings of the National Academy of Sciences* **105**, 14796–14801.

Birk, EM, Bridges, RG (1989) Recurrent fires and fuel accumulation in even-aged blackbutt (*Eucalyptus pilularis*) forests. *Forest Ecology and Management* **29**, 59-79.

Black, A, Opperman, T (2005) The fire effects planning framework. *International Journal of Wilderness* **11**, 19-20.

Boland, DJ, Brooker, MIH, Chippendale, GM, Hall, N, Hyland, BPM, Johnston, RD, Kleinig, DA, McDonald, MW, Turner, JD (2006) Forest trees of Australia. MW McDonald. CSIRO: Collingwood, Victoria. 736.

Bradstock, RA, Gill, AM (1993) Fire in semiarid, mallee shrublands-size of flames from discrete fuel arrays and their role in the spread of fire. *International Journal of Wildland Fire* **3**, 3-12.

Bradstock, RA, Williams, J, Gill, M (2002) Flammable Australia : the fire regimes and biodiversity of a continent. Cambridge University Press: New York. 462.

Bradstock, RA (2010) A biogeographic model of fire regimes in Australia: current and future implications. *Global Ecology and Biogeography* 19, 145-158.

Braithwaite, RW, Lonsdale, WM, Estbergs, JA (1989) Alien vegetation and native biota in tropical Australia: the impact of Mimosa pigra. *Biological Conservation* 48, 189-210.

Briese, DT (1996) Biological control of weeds and fire management in protected natural areas: Are they compatible strategies? *Biological Conservation* **77**, 135-141.

Brooks, ML, D'Antonio, CM, Richardson, DM, Grace, JB, Keeley, JE, Ditomaso, JM, Hobbs, RJ, Pellant, M, Pyke, D (2004) Effects of invasive alien plants on fire regimes. *BioScience* **54**, 677-688.

Brown, JK (1974) Handbook for inventorying downed woody material. USDA Forest Service Technical Report. 32.

Brown, JK, Oberheu, RD, Johnston, CM (1982) Handbook for inventorying surface fuels in the interior West. USDA Forest Service General Technical Report. 52.

Bowman, DMJS, French, BJ, Prior, LD (2014) Have plants evolved to self-immolate? *Frontiers in plant science* **5**, 1-9.

Buckley, YM, Bolker, BM, Rees, M (2007) Disturbance, invasion and re-invasion: managing the weed-shaped hole in disturbed ecosystems. *Ecology Letters* **10**, 809-817.

Burgan, RE, Rothermel, RC (1984) BEHAVE: Fire behaviour prediction and fuel modeling system. USDA Forest Service Technical Report. 134.

Burrows, ND (1994) Experimental development of a fire management model for jarrah (*Eucalyptus marginata* Donn ex Sm.) forest. PhD thesis, Australian National University, Canberra.

Burrows, ND (1999a) Fire behaviour in jarrah forest fuels. *CALMScience* **3**, 31-56.

Burrows, ND (1999b) Fire behaviour in jarrah forest fuels: 2. Field experiments. *CALMScience* **3**, 57-84.

Burrows, ND, Ward, B, Robinson, A (1991). Fire behaviour in spinifex fuels on the Gibson Desert Nature Reserve, Western Australia. *Journal of Arid Environment*, **20**, 189-204.

Burrows, ND, Ward, B, Robinson, A (2009) Fuel dynamics and fire spread in spinifex grasslands of the Western Desert. Proceedings of the Royal Society of Queensland **115**, 69.

Byram, GM (1959) Combustion of forest fuels. In: Forest Fire: Control and Use. KP Davis. New York. 686.

Callaway, RM (1997) Positive interactions in plant communities and the individualistic-continuum concept. *Oecologia* **112**, 143-149.

Callaway, RM, Aschehoug, ET (2000) Invasive plants versus their new and old neighbors: A mechanism for exotic invasion. *Science* **290**, 521-523.

Canfield, RH (1941) Application of the Line Interception Method in Sampling Range Vegetation. *Journal of Forestry* **39**, 388-394.

Cardinale, BJ, Duffy, JE, Gonzalez, A, Hooper, DU, Perrings, C, Venail, P, Narwani, A, Mace, GM, Tilman, D, Wardle, DA (2012) Biodiversity loss and its impact on humanity. *Nature* **486**, 59-67.

Carr, GW, Yugovic, JV, Robinson, KE, Pty, EH (1992) Environmental weed invasions in Victoria: conservation and management implications. Department of Conservation and Environment and Ecological Horticulture Pty Limited.

Catchpole, W (2002) Fire properties and burn patters in heterogeneous landscapes. In: Flammable Australia: the fire regimes and biodiversity of a continent. RA Bradstock, AM Gill, RJ Williams. CSIRO Publishing: Collingwood, Victoria. 488.

Chandler, C, Cheney, P, Thomas, P, Trabaud, L, Williams, D (1983) Fire effects on soil, water and air. *Fire in forestry* **1**, 171-202.

Chapin, FS, Trainor, SF, Huntington, O, Lovecraft, AL, Zavaleta, E, Natcher, DC, McGuire, AD, Nelson, JL, Ray, L, Calef, M (2008) Increasing wildfire in Alaska's boreal forest: pathways to potential solutions of a wicked problem. *BioScience* **58**, 531-540.

Cheney NP, Gould JS, Catchpole WR (1993) The Influence of Fuel, Weather and Fire Shape Variables on Fire-Spread in Grasslands. *International Journal of Wildland Fire* **3**, 31-44.

Cheney NP, Gould JS, Catchpole WR (1998) Prediction of Fire Spread in Grasslands. *International Journal of Wildland Fire* **8**, 1-13.

Cheney, N, Gould, J, Catchpole, W (1993) The Influence of Fuel, Weather and Fire Shape Variables on Fire-Spread in Grasslands. *International Journal of Wildland Fire* **3**, 31-44.

Cheney, NP (1981) Fire behaviour. In: Fire and the Australian biota. AM Gill, RH Groves, IR Noble. Australian Academy of Science: Canberra. 151-175.

Cheney, NP (1990) Quantifying bushfires. *Mathematical and Computer Modelling* **13**, 9-15.

Cheney, NP, Sullivan, A (2008) Grassfires: fuel, weather and fire behaviour. CSIRO Publishing. 160.

Cheney, NP, Gould, JS (1996) Development of Fire Behaviour Models for High-intensity Forest Fires. In: Conference on Natural Disaster Reduction. Surfers Paradise, Queensland. Barton. 165-170.

Cheney, NP, Gould, JS, Knight, I (1992). A prescribed burning guide for young regrowth forests of silvertop ash. Sydney: Research Division, Forestry Commission of New South Wales. 92.

Cheney, NP, Gould, JS, Knight, I, Wales, NS (1992) A prescribed burning guide for young regrowth forests of silvertop ash. Research Division, Forestry Commission of New South Wales Sydney.

Cheney, NP, Gould, JS, McCaw, WL, Anderson, WR (2012) Predicting fire behaviour in dry eucalypt forest in southern Australia. *Forest Ecology and Management* **280**, 120-131.

Cheney, NP, Gould, JS, McCaw, WL, Anderson, WR (2012) Predicting fire behaviour in dry eucalypt forest in southern Australia. *Forest Ecology and Management* **280**, 120-131.

Christophel, CD (1989) Evolution of the Australian flora through the Tertiary. *Plant Sysramatics and Evolution* **162**, 63-78.

Clar, CR, Chatten, LR (1954) Principles of forest fire management. Departament of Natural Resources. Division of Forestry. 274.

Cochrane, MA, Ryan, KC (2009) Fire and fire ecology: Concepts and principles Tropical Fire Ecology. Springer Berlin Heidelberg. 25-62.

Cohn, JS, Lunt, ID, Ross, KA, Bradstock, RA (2011) How do slow-growing, fire-sensitive conifers survive in flammable eucalypt woodlands? *Journal of Vegetation Science* **22**, 425-435.

Coley, PD (1988) Effects of plant growth rate and leaf lifetime on the amount and type of anti-herbivore defense. *Oecologia* **74**, 531-536.

Cook, GD, Dias, L (2006) It was no accident: deliberate plant introductions by Australian government agencies during the 20th century. *Australian Journal of Botany* **54**, 601-625.

Cowling, RM, Lamont, BB (1998) On the nature of gondwanan species flocks: diversity of proteaceae in Mediterranean South-Western Australia and South Africa. *Australian Journal of Botany* **46**, 335-355.

Crossman, ND (2002) The impact of the European olive (*Olea europaea* L. subsp. *europaea*) on grey box (*Eucalyptus microcarpa* Maiden) woodland in South Australia. *Plant Protection Quarterly* **17**, 140-146.

Cruz, MG, Alexander, ME (2010) Assessing crown fire potential in coniferous forests of western North America: a critique of current approaches and recent simulation studies. *International Journal of Wildland Fire* **19**, 377-398.

Cruz, MG, Alexander, ME (2013) Uncertainty associated with model predictions of surface and crown fire rates of spread. *Environmental Modelling & Software* **47**, 16-28.

Cruz, MG, Fernandes, PM (2008) Development of fuel models for fire behaviour prediction in maritime pine (*Pinus pinaster* Ait.) stands. *International Journal of Wildland Fire* **17**, 194-204.

Cruz, MG, Matthews, S, Gould, J, Ellis, PF, Henderson, M, Knight, I, Watters, J (2010) Fire Dynamics in Mallee-Heath: Fuel, Weather and Fire Behaviour Prediction on South Australian Semi-Arid Shrublands. Bushfire Cooperative Research Centre, East Melbourne, Victoria. 134.

Cruz, MG, McCaw, WL, Anderson, WR, Gould, JS (2013) Fire behaviour modelling in semi-arid mallee-heath shrublands of southern Australia. *Environmental Modelling & Software* **40**, 21-34.

Countryman, CM (1972) The fire environment concept. Pacific Southwest Forest and Range Experiment Station: Berkeley.

Cuneo, P, Jacobson, CR, Leishman, MR (2009) Landscape-scale detection and mapping of invasive African Olive (*Olea europaea* L. ssp. *cuspidata* Wall ex G. Don Ciferri) in NSW Sydney, Australia using satellite remote sensing. *Applied Vegetation Science* **12**, 145-154.

Cuneo, P, Leishman, MR (2006) African Olive (*Olea europaea* subsp. *cuspidata*) as an environmental weed in eastern Australia: a review. *Cunninghamia* **9**, 545-577.

Cuneo, P, Leishman, MR (2012) Ecological impacts of invasive African olive (*Olea europaea* ssp. *cuspidata*) in Cumberland Plain Woodland, Sydney, Australia. *Austral Ecology* **38**, 103-110.

Cunningham, SA, Summerhayes, B, Westoby, M (1999) Evolutionary Divergences in Leaf Structure and Chemistry, Comparing Rainfall and Soil Nutrient Gradients. *Ecological Monographs* **69**, 569-588.

D'Antonio, CM, Tunison, JT, Loh, RK (2000) Variation in the impact of exotic grasses on native plant composition in relation to fire across an elevation gradient in Hawaii. *Austral Ecology* **25**, 507-522.

Daehler, CC (2003) Performance Comparisons of Co-Occurring Native and Alien Invasive Plants: Implications for Conservation and Restoration. *Annual Review of Ecology, Evolution, and Systematics* **34**, 183-211.

D'Antonio, CM, Vitousek, PM (1992) Biological Invasions by Exotic Grasses, the Grass/Fire Cycle, and Global Change. *Annual Review of Ecology and Systematics* **23**, 63-87.

Dawson, W, Burslem, DFRP, Hulme, PE (2009) Factors explaining alien plant invasion success in a tropical ecosystem differ at each stage of invasion. *Journal of Ecology* **97**, 657-665.

De Groot, WJ (1993) Examples of fuel types in the Canadian Forest Fire Behavior Prediction (FBP) System. Forestry Canada Northwest Region Northern Forestry Centre. 5320-122.

De Lillis, M, Bianco, PM, Loreto, F (2009) The influence of leaf water content and isoprenoids on flammability of some Mediterranean woody species. *International Journal of Wildland Fire* **18**, 203-212.

de Magalhaes, RMQ, Schwilk, DW (2012) Leaf traits and litter flammability: evidence for non-additive mixture effects in a temperate forest. *Journal of Ecology* **100**, 1153-1163.

DeBano, LF, Neary, DG, Ffolliott, PF (1998) Fire effects on ecosystems. John Wiley & Sons. 333.

Deeming, JE, Burgan, RE, Cohen, JD (1977) The National Fire-Danger Rating System--1978. *USDA Forest Service General Technical Report INTUS (USA). no. 39.*

Department of Primary Industries (2008) NSW Invasive Species Plan. NSW Department of Primary Industries. Orange, NSW. 36.

Di Castri, F, Goodall, DW, Specht, RL (1981) Mediterranean-type shrublands. F Di Castri, DW Goodall, RL Specht. Elsevier. Amsterdam. 643.

Diaz, S, Hodgson, JG, Thompson, K, Cabido, M, Cornelissen, JHC, Jalili, A, Montserrat-Marti, G, Grime, JP, Zarrinkamar, F, Asri, Y (2004) The plant traits that drive ecosystems: evidence from three continents. *Journal of Vegetation Science* **15**, 295-304.

Dibble, AC, White, RH, Lebow, PK (2007) Combustion characteristics of north-eastern USA vegetation tested in the cone calorimeter: invasive versus non-invasive plants. *International Journal of Wildland Fire* **16**, 426-443.

Dickinson, KJM, Kirkpatrick, JB (1985) The Flammability and Energy Content of Some Important Plant Species and Fuel Components in the Forests of Southeastern Tasmania. *Journal of Biogeography* **12**, 121-134.

Dimitrakopoulos, AP (2001) Thermogravimetric analysis of Mediterranean plant species. *Journal of Analytical and Applied Pyrolysis* **60**, 123-130.

Dimitrakopoulos, AP (2002) Mediterranean fuel models and potential fire behaviour in Greece. *International Journal of Wildland Fire* **11**, 127-130.

Dimitrakopoulos, AP, Mitsopoulos, ID, Kaliva, A (2013) Comparing flammability traits among fire-stricken (low elevation) and non fire-stricken (high elevation) conifer forest species of Europe: A test of the Mutch hypothesis. *Forest Systems* **22**, 134-137.

Dimitrakopoulos, AP, Papaioannou, K (2001) Flammability Assessment of Mediterranean Forest Fuels. *Fire Technology* **37**, 143-152.

Donovan, LA, Maherali, H, Caruso, CM, Huber, H, de Kroon, H (2011) The evolution of the worldwide leaf economics spectrum. *Trends in Ecology & Evolution* **26**, 88-95.

Drenovsky, RE, Grewell, BJ, D'Antonio, CM, Funk, JL, James, JJ, Molinari, N, Parker, IM, Richards, CL (2012) A functional trait perspective on plant invasion. *Annals of Botany.* MCS 100.

Drysdale, D (2011) An introduction to fire dynamics. John Wiley & Sons. 574.

Dupuy, J-L, Larini, M (2000) Fire spread through a porous forest fuel bed: A radiative and convective model including fire-induced flow effects. *International Journal of Wildland Fire* **9**, 155-172.

Emms, J, Virtue, JG, Preston, C, Bellotti, WD (2005) Legumes in temperate Australia: A survey of naturalisation and impact in natural ecosystems. *Biological Conservation* **125**, 323-333.

Environment Protection and Biodiversity Conservation Act (EPBC) (1999). Australian Government, Office of Legislative Drafting and Publishing (OLDP), Canberra.

Etlinger, MG, Beall, FC (2005) Development of a laboratory protocol for fire performance of landscape plants. *International Journal of Wildland Fire* **13**, 479-488.

Fang, JB, Steward, FR (1969) Flame spread through randomly packed fuel particles. *Combustion and Flame* **13**, 392-398.

Fernandes, PM (2009) Combining forest structure data and fuel modelling to classify fire hazard in Portugal. *Annals of Forest Science* **66**, 1-9.

Fernandes, PM, Botelho, HS (2003) A review of prescribed burning effectiveness in fire hazard reduction. *International Journal of Wildland Fire* **12**, 117-128.

Fernandes, PM, Cruz, MG (2012) Plant flammability experiments offer limited insight into vegetation–fire dynamics interactions. *New Phytologist* **194**, 606-609.

Fernández-Escobar, R, Moreno, R, García-Creus, M (1999) Seasonal changes of mineral nutrients in olive leaves during the alternate-bearing cycle. *Scientia Horticulturae* **82**, 25-45.

Fernandez-Escobar, R, Moreno, R, Sanchez-Zamora, MA (2004) Nitrogen dynamics in the olive bearing shoot. *HortScience* **39**, 1406-1411.

Fine, PVA (2002) The invasibility of tropical forests by exotic plants. *Journal of Tropical Ecology* **18**, 687-705.

Fine, PVA (2002) The invasibility of tropical forests by exotic plants. *Journal of Tropical Ecology* **18**, 687-705.

Finkral, AJ, Evans, AM (2008) The effects of a thinning treatment on carbon stocks in a northern Arizona ponderosa pine forest. *Forest Ecology and Management* **255**, 2743-2750.

Finney, MA (1998) FARSITE: fire area simulator-model. *Model development and evaluation.* USDA Forest Service Technical Report. 50.

Fisher, JL, Loneragan, WA, Dixon, K, Delaney, J, Veneklaas, EJ (2009) Altered vegetation structure and composition linked to fire frequency and plant invasion in a biodiverse woodland. *Biological Conservation* **142**, 2270-2281.

Florence, RG, Lamb, D (1975) Ecosystem processes and the management of radiata pine forests on sand dunes in South Australia. In Proceedings of the Ecological Society of Australia.

Flory, S, Clay, K (2010) Non-native grass invasion alters native plant composition in experimental communities. *Biological Invasions* **12**, 1285-1294.

Floyd, AG (1966) Effect of fire upon weed seeds in the wet Sclerophyll forests of Northern New South Wales. *Australian Journal of Botany* **14**, 243-256.

Fogarty, G, Facelli, JM (1999) Growth and competition of *Cytisus scoparius*, an invasive shrub, and Australian native shrubs. *Plant Ecology* **144**, 27-35.

Fons, WL (1946) Analysis of fire spread in light forest fuels. *Journal of Agriculture* **27**, 93-121.

Forbes, LK (1997) A two-dimensional model for large-scale bushfire spread. *The Journal of the Australian Mathematical Society. Applied Mathematics* **39**, 171-194.

Forestry Canada Science and Sustainable Development Directorate (FCSSDD) (1992) Development and structure of the Canadian Forest Fire Behavior prediction System. Information report ST-X-3. Forestry Canada Fire Danger Group. Canada. 66.

Fosberg MA (1970) Drying Rates of Heartwood Below Fiber Saturation. *Forest Science* **16**, 57-63.

Francesca Cotrufo, M, Ineson, P, Derek Roberts, J (1995) Decomposition of birch leaf litters with varying C-to-N ratios. *Soil Biology and Biochemistry* **27**, 1219-1221.

Frandsen, WH (1971) Fire spread through porous fuels from the conservation of energy. *Combustion and Flame* **16**, 9-16.

Frandsen, WH (1973) Effective heating of fuel ahead of a spreading fire. US Department of Agriculture, Forest Service, Research Paper INT-140. Intermountain Forest and Range Experiment Station. Ogden, Utah. 16.

Frost, PG, Robertson, F (1987) The ecological effects of fire in savannas. In: Determinants of tropical savannas. BH Walker. IRL Press Limited: Oxford.

Funk, JL, Vitousek, PM (2007) Resource-use efficiency and plant invasion in low-resource systems. *Nature* **446**, 1079-1081.

Gaertner, M, Biggs, R, Te Beest, M, Hui, C, Molofsky, J, Richardson, DM (2014) Invasive plants as drivers of regime shifts: identifying high-priority invaders that alter feedback relationships. *Diversity and Distributions* **20**, 733-744.

Gaertner, M, Den Breeyen, A, Hui, C, Richardson, DM (2009) Impacts of alien plant invasions on species richness in Mediterranean-type ecosystems: a meta-analysis. *Progress in Physical Geography* **33**, 319-338.

Ganteaume A, Lampin-Maillet C, Guijarro M, Hernando C, Jappiot M, Fonturbel T, Pérez-Gorostiaga P, Vega JA (2009) Spot fires: fuel bed flammability and capability of firebrands to ignite fuel beds. *International Journal of Wildland Fire* **18**, 951-969.

Ganteaume, A, Jappiot, M, Lampin, C, Guijarro, M, Hernando, C (2013) Flammability of Some Ornamental Species in Wildland–Urban Interfaces in Southeastern France: Laboratory Assessment at Particle Level. *Environmental Management* **52**, 467-480.

Gartner, TB, Cardon, ZG (2004) Decomposition dynamics in mixed-species leaf litter. *Oikos* **104**, 230-246.

Gibbons, P, Boak, M (2002) The value of paddock trees for regional conservation in an agricultural landscape. *Ecological Management & Restoration* **3**, 205-210.

Gill, AM, Groves, R.H., Noble, I.R. (1981) Fire and Australian Biota. Australian Academy of Sciences: Canberra. 34.

Gill, AM, Moore, PH (1996) Ignitibility of leaves of Australian plants. CSIRO. 34.

Gill, AM, Moore, PHR, Williams, RJ (1996) Fire weather in the wet-dry tropics of the World Heritage Kakadu National Park, Australia. *Australian Journal of Ecology* **21**, 302-308.

Gill, MA, Catling, PC (2002) Fire Regimes and biodiversity of forested landscapes of southern Australia. In: Flammable Australia: the fire regimes and biodiversity of a continent. RA Bradstock, JE Williams, MA Gill. Cambridge University Press: Cambridge. 351-372.

Gill, MA, Zylstra, P (2005) Flammability of Australian forests. *Australian Forestry* **68**, 87-93.

Gillon, D, Hernando, C, Valette, JC, Joffre, R (1997) Fast estimation of the calorific values of forest fuels by near infrared-reflectance spectroscopy. *Canadian Journal of Forest Research* **27**, 760-765.

Goodall, J, Witkoski, ETF, Morris, CD, Henderson, L (2011) Are environmental factors important facilitators of pompom weed (*Campuloclinium macrocephalum*) invasion in South African rangelands? *Biological Invasions* 2217-2231.

Gosper, CR, Yates CJ, Prober SM, Wiehl, G (2014) Application and validation of visual fuel hazard assessments in dry Mediterranean-climate woodlands. *International Journal of Wildland Fire* **23**, 385-393.

Gould, J, Cheney, NP, McCaw, WL, Anderson, W (2013) Predicting fire behaviour in dry eucalypt forest. In: CSIRO Bushfire behaviour science symposium. CSIRO. Camberra. 67.

Gould, JS (1991) Validation of the Rothermel fire spread model and related fuel parameters in grassland fuels. In: Conference on Bushfire Modelling and Fire Danger Rating Systems. NP, Cheney, AM, Gill. 51-64.

Gould, JS, Cheney, NP, McCaw, L, Anderson, W (2013) Predicting fire behaviour in dry eucalypt forest. In: CSIRO Bushfire behaviour science symposium. CSIRO. Camberra. 67.

Gould, JS, Knight, I, Sullivan, AL (1997) Physical modelling of leaf scorch height from prescribed fires in young Eucalyptus sieberi regrowth forests in south-eastern Australia. *International Journal of Wildland Fire* **7**, 7-20.

Gould, JS, McCaw, WL, Cheney, NP (2011) Quantifying fine fuel dynamics and structure in dry eucalypt forest (*Eucalyptus marginata*) in Western Australia for fire management. *Forest Ecology and Management* **262**, 531-546.

Gould, JS, McCaw, WL, Cheney, NP, Ellis PF, Matthews S (2007b) Field guide - Fuel Assessment and Fire Behaviour Prediction in Dry Eucalypt Forest. Department of Environment and Conservation. Camberra ACT. 83.

Gould, JS, McCaw, WL, Cheney, NP, Ellis, PF, Knight, IK, Sullivan, AL (2007a) Project Vesta - Fire in dry Eucalypt Forest: Fueal structure, fuel dinamics and fire behaviour. Ensis-CSIRO. Camberra ACT. 218.

Government of Queensland (2014) Environmental weeds of Australia for Biosecurity Queensland. University of Queensland. Retrieved July 01, 2014, from http://keyserver.lucidcentral.org/weeds/data/03030800-0b07-490a-8d04-0605030c0f01/media/Html/Index.htm

Government of South Australia (2001) Declared plant policy - Olive (*Olea europea*). Government of South Australia, South Australia. 5.

Government of South Australia (2004) Feral Olives in South Australia. Department of Water, Land and Biodiversity Conservation . South Australia. 17.

Grabner, K, Dwyer J, Cutter, B (1997) Validation of BEHAVE fire behavior predictions in oak savannas using five fuel models. In: Proceedings of the 11th Central Hardwood Forest Conference. Department of Agriculture, Forest Service, North Central Forest Experiment Station. 202-215.

Graça, MA, Bärlocher, F, Gessner, MO (2005) Methods to Study Litter Decomposition - A practical guide. MA Graça. Springer. 329.

Grace, JB (1998) Can prescribed fire save the endangered coastal prairie ecosystem from Chinese tallow invasion. *Endangered Species Update* **15**, 70-76.

Graham, RT, McCaffrey, S, Jain, TB (2004) Science basis for changing forest structure to modify wildfire behavior and severity. USDA Forest Service Technical Report. 46.

Griffin, GF, Friedel, MH (1984) Effects of fire on central Australian rangelands. I Fire and fuel characteristics and changes in herbage and nutrients. *Australian Journal of Ecology* **9**, 381-393.

Grishin, AM (1997) Mathematical modeling of forest fires and new methods of fighting them. Publishing house of the Tomsk State University. 451.

Grooves, RH (1991) Biogeography of mediterranean invasions. Cambridge University Press: Oakleigh, Victoria. 504.

Groves, RH (2002) The impacts of alien plants in Australia. In: Biological invasions: economic and environmental costs of alien plant, animal, and microbe species. D Pimentel. CRC Press: Washington D.C. 11-24.

Groves, RH (2006) Are some weeds sleeping? Some concepts and reasons. *Euphytica* **148**, 111-120.

Groves, RH, Hosking, JR, Batianoff, GN, Cooke, DA, Cowie, ID, Johnson, RW, Keighery, GJ, Lepschi, BJ, Mitchell, AA, Moerkerk, M, Randall, RP, Rozefelds, AC, Walsh, NG, Waterhouse, BM (2003) Weed categories for natural and agricultural ecosystem management. Bureau of Rural Sciences: Camberra. 194.

Groves, RH, Lonsdale, M, Boden, R (2005). Jumping the garden fence: invasive garden plants in Australia and their environmental and agricultural impacts. WWF-Australia: Sydney. 173.

Groves, RH, Willis, AJ (1999) Environmental weeds and loss of native plant biodiversity: some Australian examples. *Australian Journal of Environmental Management* **6**, 164-171.

Gul, S, Whalen, JK, Ellis, BE, Grayston, SJ (2012) Plant residue chemistry impacts soil processes and microbial community structure: A study with Arabidopsis thaliana cell wall mutants. *Applied Soil Ecology* **60**, 84-91.

Gurevitch, J, Padilla, DK (2004) Are invasive species a major cause of extinctions? *Trends in Ecology & Evolution* **19**, 470-474.

Harmon, ME, Nadelhoffer, KJ, Blair, JM (1999) Measuring decomposition, nutrient turnover, and stores in plant litter. *Standard soil methods for long-term ecological research.* Oxford University Press, New York 202-240.

Harrington, S (2005) Measuring Forest Fuels: An Overview of Methodologies - Implication for fuel management. 1-27.

He, T, Lamont, BB, Downes, KS (2011) Banksia born to burn. *New Phytologist* **191**, 184-196.

Hessburg, PF, Reynolds, KM, Keane, RE, James, KM, Salter, RB (2007) Evaluating wildland fire danger and prioritizing vegetation and fuels treatments. *Forest Ecology and Management* **247**, 1-17.

Higgins, SI, Richardson, DM (1996) A review of models of alien plant spread. *Ecological Modelling* **87**, 249-265.

Hill, RS (2004) Origins of the southeastern Australian vegetation. *Philosophical Transactions of the Royal Society of London* 1537-1549.

Hill, SJ, Tung, PJ, Leishman, MR (2005) Relationships between anthropogenic disturbance, soil properties and plant invasion in endangered Cumberland Plain Woodland, Australia. *Austral Ecology* **30**, 775-788.

Hines, F, Tolhurst, KG, Wilson, AAG, McCarthy, GJ (2010) Overall fuel hazard assessment guide, fourth edition. Melbourne, Victoria. 47.

Hirsch, KG (1996) Canadian forest fire behavior prediction (FBP) system: user's guide. atural Resources Canada, Canadian Forest Service, Northern Forestry Centre, Edmonton, Alberta. Special Report 7. 122.

Hobbs, RJ, Huenneke, LF (1992) Disturbance, Diversity, and Invasion: Implications for Conservation. *Conservation Biology* **6**, 324-337.

Hollis, JJ, Matthews, S, Ottmar, RD, Prichard, SJ, Slijepcevic, A, Burrows, ND, Ward, B, Tolhurst, KG, Anderson, WR, Gould, JS (2010) Testing woody fuel consumption models for application in Australian southern eucalypt forest fires. *Forest Ecology and Management* **260**, 948-964.

Hopper, SD, Gioia, P (2004) The Southwest Australian Floristic Region: Evolution and Conservation of a Global Hot Spot of Biodiversity. *Annual Review of Ecology, Evolution, and Systematics* **35**, 623-650.

Hulme, PE (2009) Trade, transport and trouble: managing invasive species pathways in an era of globalization. *Journal of Applied Ecology* **46**, 10-18.

Humphries, SE, Groves, RH, Mitchell, DS (1991) Plant invasions of Australian ecosystems: a status review and management directions. Australian National Parks and Wildlife Service. 165.

Interim Biogeographic Regionalisation for Australia (IBRA7) (2014) Australia's bioregions (IBRA). Retrieved April 29, 2014 from http://www.environment.gov.au/topics/land/national-reserve-system/science-maps-and-data/australias-bioregions-ibra#ibra.

Janssens, ML (2000) Heat Release Rate (HRR). In: Proceedings of an International Workshop (NISTIR 6527), Measurement Needs for Fire Safety. TJ Ohlemiller, EL Johnson, RG Gann. Gaithersburg. 186-206.

Jones, R (1969) Fire-stick farming. *Australian Natural History* **16**, 224-228.

Jurskis, V (2005) Decline of euclypt forests as a consequence of unnatural fire regimes. *Australian Forestry* **68**, 257-262.

Jurskis, V, Bridges, B, de Mar, P (2003) Fire management in Australia: the lessons of 200 years. In In Joint Australia and New Zealand Institute of Forestry Conference Proceedings **27**, 353-368.

Keane, RE (2013) Describing wildland surface fuel loading for fire management: a review of approaches, methods and systems. *International Journal of Wildland Fire* **22**, 51-62.

Keeley, JE (2009) Fire intensity, fire severity and burn severity: a brief review and suggested usage. *International Journal of Wildland Fire* **18**, 116-126.

Keeley, JE, Pausas, JG, Rundel, PW, Bond, WJ, Bradstock, RA (2011) Fire as an evolutionary pressure shaping plant traits. *Trends in Plant Science* **16**, 406-411.

Keeley, JE, Zedler, PH (1998) Evolution of life histories in Pinus. In: Ecology and Biogeografy of Pinus. DM Richardson. Cabridge University Press: Cambridge. 219-250.

Keith, DA, McCaw, L, Whelan, RJ (2002) Fire regimes in Australian heathlands and their effects on plants and animals. In: Flammable Australia: The Fire Regimes and Biodiversity of a Continent. R Bradstock, J Williams, AM Gill. Cambridge University Press. 199-237.

Keith, DA, Wales, NS (2004) Ocean shores to desert dunes: the native vegetation of New South Wales and the ACT. Department of Environment and Conservation NSW. 353.

Kent, M (2011) Vegetation description and data analysis: a practical approach. John Wiley & Sons. New York. 432.

Kershaw, PA, Clark, JS, Gill, AM, D`Costa, DM (2002) A history of fire in Australia. In: Flammable Australia: The fire regimes and Biodiversity of a continent. RA Bradstock, JE Williams, MA Gill. Cambridge University Press. 3-25.

Klemmedson, JO (1992) Decomposition and nutrient release from mixtures of Gambel oak and ponderosa pine leaf litter. *Forest Ecology and Management* **47**, 349-361.

Kloot, PM (1991) Invasive plants of southern Australia In Biogeografy of Mediterranean invasions. Groves RH. Cambridge University Press: New York. 115-130.

Kozlowski TT (2012) Fire and Ecosystems. Academic Press. University of Michigan. 542.

Lake, JC, Leishman, MR (2004) Invasion success of exotic plants in natural ecosystems: the role of disturbance, plant attributes and freedom from herbivores. *Biological Conservation* **117**, 215-226.

Leishman, MR, Haslehurst, T, Ares, A, Baruch, Z (2007) Leaf trait relationships of native and invasive plants: community-and global-scale comparisons. *New Phytologist* **176**, 635-643.

Liao, C, Peng, R, Luo, Y, Zhou, X, Wu, X, Fang, C, Chen, J, Li, B (2008) Altered ecosystem carbon and nitrogen cycles by plant invasion: A meta-analysis. *New Phytologist* **177**, 706-714.

Linn, RR, Harlow, FH (1997) FIRETEC: a transport description of wildfire behavior. Los Alamos National Lab. NM United States.

Lockwood, JL, Hoopes, MF, Marchetti, MP (2007) Invasion Ecology. Wiley-Blackwell. 466.

Lockwood, JL, Hoopes, MF, Marchetti, MP (2013) Invasion ecology. 2 edition. Wiley-Blackwell. 466.

Lonsdale, WM (1994) Inviting trouble: Introduced pasture species in northern Australia. *Australian Journal of Ecology* **19**, 345-354.

Loveday, J (1974) Methods for analysis of irrigated soils. Farnham Royal, Buckinghamshire: Commonwealth Agricultural Bureaux. 208

Luke, RH, McArthur, AG (1978) Bushfires in Australia. AGPS. Australia: Canberra.

Lumaret R, Ouazzani N (2001) Plant genetics: Ancient wild olives in Mediterranean forests. *Nature* **413**, 700-700.

Mack, MC, D'Antonio, CM (1998) Impacts of biological invasions on disturbance regimes. *Trends in Ecology & Evolution* **13**, 195-198.

Mack, RN, Lonsdale, WM (2001) Humans as global plant dispersers: Getting more than we bargained for. *BioScience* **51**, 95-102.

Mackensen, J, Bauhus, J, Webber, E (2003) Decomposition rates of coarse woody debris—a review with particular emphasis on Australian tree species. *Australian Journal of Botany* **51**, 27-37.

Madrigal, J, Hernando, C, Guijarro, M (2013) A new bench-scale methodology for evaluating the flammability of live forest fuels. *Journal of Fire Sciences* **31**, 131-142.

Madrigal, J, Hernando, C, Guijarro, M, Díez, C, Marino, E, De Castro, AJ (2009) Evaluation of Forest Fuel Flammability and Combustion Properties with an Adapted Mass Loss Calorimeter Device. *Journal of Fire Sciences* **27**, 323-342.

Mandle, L, Bufford J, Schmidt, I, Daehler, C (2011) Woody exotic plant invasions and fire: reciprocal impacts and consequences for native ecosystems. *Biological Invasions* **13**, 1815-1827.

Marino, E, Madrigal, J, Guijarro, M, Hernando, C, Díez, C, Fernández, C (2010) Flammability descriptors of fine dead fuels resulting from two mechanical treatments in shrubland: a comparative laboratory study. *International Journal of Wildland Fire* **19**,

Marsden-Smedley, JB, Catchpole, WR (1995) Fire behaviour modelling in Tasmanian buttongrass moorlands. II. Fire behaviour. *International Journal of Wildland Fire* **5**, 215-228.

Martin, RE, Gordon, DA, Gutierrez, MA, Lee, DS, Molina, DM, Schroeder, RA, Sapsis, DB, Stephens, SL, Chambers, M (1994) Assessing the flammability of domestic and wildland vegetation. In: Proceedings of the 12th conference on fire and forest meteorology. 26-28.

Matthews, S (2010) Effect of drying temperature on fuel moisture content measurements. *International Journal of Wildland Fire* **19**, 800-802.

Matthews, S, Gould J, McCaw, L (2010) Simple models for predicting dead fuel moisture in eucalyptus forests. *International Journal of Wildland Fire* **19**, 459-467.

McAlpine, RS (1995) Testing the Effect of Fuel Consumption on Fire Spread Rate. *International Journal of Wildland Fire* **5**, 143-152.

McArthur, AG (1962) Control burning in eucalypt forests. Commonwealth of Australia, Forestry and Timber Bureau.

McArthur, AG (1966) Weather and grassland fire behaviour. Forestry and Timber Bureau, Department of national Development, Commonwealth of Australia.

McArthur, AG (1967) Fire behaviour in eucalypt forests. Leaflet Number 107 Commonwealth of Australia Forestry and Timber Bureau.

McArthur, AG (1973) Forest Fire Danger Meter Mk. 5. Forest Research Institute, Forestry and Timber Bureau, Canberra.

McCarthy, GJ, Tolhurst, KG, Chatto, K (1999) Overall fuel hazard guide. Melbourne

McCaw, LW, Gould, JS, Cheney, NP, Ellis, PFM, Anderson, WR (2012) Changes in behaviour of fire in dry eucalypt forest as fuel increases with age. *Forest Ecology and Management* **271**, 170-181.

McCaw, WL (1997) Predicting fire spread in Western Australian mallee-heath shrublands. University of New South Wales. PhD thesis. 235.

McCaw, WL, Gould JS, Cheney, NP (2008) Existing fire behaviour models under-predict the rate of spread of summer fires in open jarrah (Eucalyptus marginata) forest. *Australian Forestry* **71**, 16-26.

McLoughlin, S (2001) The breakup history of Gondwana and its impact on pre-Cenozoic floristic provincialism. *Australian Journal of Botany* **49**, 271-300.

Melillo, JM, Aber, JD, Muratore, JF (1982) Nitrogen and lignin control of hardwood leaf litter decomposition dynamics. *Ecology* **63**, 621-626.

Mell, W, Jenkins, MA, Gould, J, Cheney, P (2007) A physics-based approach to modelling grassland fires. *International Journal of Wildland Fire* **16**, 1-22.

Merrill, DF, Alexander, ME (1987) Glossary of forest fire management terms. National Research Council of Canada, Committee for Forest Fire Management: Ottawa, Ontario.

Miranda, HS, Rocha e Silva, EP, Miranda, AC (1996) Comportamento do fogo em queimadas de campo sujo. In: Impacto de queimadas em areas de Cerrado e Restinga. HS Miranda, CH Saito, BFS Dias. Brasilia, DF. 1-10.

Mladovan, L (1998) Avian Dispersal of the European Olive *Olea Europaea* Especially by the Common Starling *Sturnus vulgaris*: Ecological Implications for Weed Management. University of Adelaide. PhD thesis.

Moore, PF (1986) Comparison of the Forest Fire Danger Meter Mk. 5 and the BEHAVE Fire Behavior Prediction System in a Dry Eucalypt Forest. University of Montana. PhD thesis.

Morgan, A, Carthew, SM, Sedgley, M (2002) Breeding system, reproductive efficiency and weed potential of A. baileyana. *Australian Journal of Botany* **50**, 357-364.

Morgan, JW (1999) Defining grassland fire events and the response of perennial plants to annual fire in temperate grasslands of south-eastern Australia. *Plant Ecology* **144**, 127-144.

Morvan, D, Larini, M (2001) Modeling of one dimensional fire spread in pine needles with opposing air flow. *Combustion Science and Technology* **164**, 37-64.

Mudrick, DA, Hoosein, M, Hicks Jr, RR, Townsend, EC (1994) Decomposition of leaf litter in an Appalachian forest: effects of leaf species, aspect, slope position and time. *Forest Ecology and Management* **68**, 231-250.

Mullett, T, Simmons, D (1995) Ecological impacts of the environmental weed sweet pittosporum (Pittosporum undulatum Vent.) in dry sclerophyll forest communities, Victoria. *Plant Protection Quarterly* **10**, 131-138.

Mulvaney, DJ, Kamminga, J (1999) Prehistory of Australia. Mulvaney, J and Kamminga, J. Smithsonian Institution Press Washington, D.C. 480.

Munhoz, CBR, Felfili, JM (2005) Fenologia do estrato herbáceo-subarbustivo de uma comunidade de campo sujo na Fazenda Água Limpa no Distrito Federal, Brasil. Acta Botanica Brasilica **19**, 979-988

Murray, BR, Hardstaff, LK, Phillips, ML (2013) Differences in Leaf Flammability, Leaf Traits and Flammability-Trait Relationships between Native and Exotic Plant Species of Dry Sclerophyll Forest. *PloS one* **8**, e79205.

Nakane, K (1995) Soil carbon cycling in a Japanese cedar (*Cryptomeria japonica*) plantation. *Forest Ecology and Management* **72**, 185-197.

Noble, IR, Gill, AM, Bary, GAV (1980) McArthur's fire-danger meters expressed as equations. *Australian Journal of Ecology* **5**, 201-203.

Noble, IR, Gill, AM, Bary, GAV (1980) McArthur's fire-danger meters expressed as equations. *Australian Journal of Ecology* **5**, 201-203.

Noble, JC (1991) Behaviour of a Very Fast Grassland Wildfire on the Riverine Plain of Southeastern Australia. *International Journal of Wildland Fire* **1**, 189-196.

Noble, JC (1991) Behaviour of a Very Fast Grassland Wildfire on the Riverine Plain of Southeastern Australia. *International Journal of Wildland Fire* **1**, 189-196.

Norris, MD, Avis, PG, Reich, PB, Hobbie, SE (2013) Positive feedbacks between decomposition and soil nitrogen availability along fertility gradients. *Plant and Soil* **367**, 347-361.

Nature Conservation Act (1980). ACT Parliamentary Counsel, Australian Capitol Territory. 142.

Noxious Weeds Act (1993). New South Wales Government, Parliamentary Counsel's Office. NSW.

NSW Department of Primary Industries (2014) Weeds definitions & FAQ. Retrieved 11, August, 2014, from http://www.dpi.nsw.gov.au/agriculture/pests-weeds/weeds/definition.

NSW National Parks and Wildlife Service (NPWS) (2002) Interpretation Guidelines for the Native Vegetation Maps of the Cumberland Plain Western Sydney: Final Edition. NSW NPWS, Hurstville. 126.

NSW Scientific Committee (2010). Invasion of native plant comunities by African olive *Olea europea* L. subsp. *cuspidata* (Wall ex G.Don Ciferri) - key threatening process listing. Retrieved 11, August, 2014, from http://www.environment.nsw.gov.au/determinations/africanoliveFD.htm.

NWS Rural Fire service (2003) Standards for low intensity bush fire hazard reduction burning. 16.

NWS Rural Fire service (2006) Bushfire Environmental Assessment Code for New South Wales. 28.

O'Connell, AM, Menage, P (1983) Decomposition of litter from three major plant species of jarrah (Eucalyptus marginata Donn ex Sm.) forest in relation to site fire history and soil type. *Australian Journal of Ecology* **8**, 277-286.

O'Donnell, J, Gallagher, RV, Wilson, PD, Downey, PO, Hughes, L, Leishman, MR (2012) Invasion hotspots for non-native plants in Australia under current and future climates. *Global Change Biology* **18**, 617-629.

Office of Environment and Heritage (2014) Climate, physical features and habitats. Retrieved 12, August, 2014, from http://www.rbgsyd.nsw.gov.au/science/Evolutionary_Ecology_Research/Ecology_of_Cumberland_Plain_Woodland/woodland_at_mount_annan/conservation_research_woodland.

Olson, JS (1963) Energy storage and the balance of producers and decomposers in ecological systems. *Ecology* **44**, 322-331.

Ormeno, E, Cespedes, B, Sanchez, IA, Velasco-García, A, Moreno, JM, Fernandez, C, Baldy, V (2009) The relationship between terpenes and flammability of leaf litter. *Forest Ecology and Management* **257**, 471-482.

Ottmar, RD, Sandberg, DV, Prichard, SJ, Riccardi, CL (2003) Fuel characteristic classification system. In: Presentation at the 2nd International Wildland Fire Ecology and Fire Management Congress.

Parsons, RF (1994) Eucalypt scrubs and shrublands. In: Australian Vegetation. RH Grooves. Cambridge University Press: Cambridge. 291-319.

Paton, DC, Tucker, JR, Paton, JB, Paton, PA (1988) Avian vectors of the seeds of the European Olive Olea europaea. *South Australian Ornithologist* **30**, 158-159.

Pauchard, A, García R, Peña, E, González, C, Cavieres, L, Bustamante, R (2008) Positive feedbacks between plant invasions and fire regimes: Teline monspessulana (L.) K. Koch (Fabaceae) in central Chile. *Biological Invasions* **10**, 547-553.

Pausas, JG, Alessio, GA, Moreira, B, Corcobado, G (2012) Fires enhance flammability in *Ulex parviflorus*. *New Phytologist* **193**, 18-23.

Pausas, JG, Bradstock, RA, Keith, DA, Keeley, JE (2004) Plant fuctional traits in relation to fire in crown-fire ecosystems. *Ecology* 85, 1085-1100.

Peet, GB (1965) A fire danger rating and controlled burning guide for the Northern Jarrah (*Eucalyptus Marginata*) forest of Western Australia. Western Australia Forests Department: Perth.

Pell, AS, Tidemann, CR (1997) The impact of two exotic hollow-nesting birds on two native parrots in savannah and woodland in eastern Australia. *Biological Conservation* **79**, 145-153.

Pennington, RT, Lavin, M, Oliveira-Filho, A (2009) Woody Plant Diversity, Evolution, Ecology in the Tropics: Perspectives from Seasonally Dry Tropical Forests. *Annual Review of Ecology, Evolution, and Systematics* **40**, 21.

Pérez-Harguindeguy, N, Díaz, S, Garnier, E, Lavorel, S, Poorter, H, Jaureguiberry, P, Bret-Harte, MS, Cornwell, WK, Craine, JM, Gurvich, DE (2013) New handbook for standardised measurement of plant functional traits worldwide. *Australian Journal of Botany* **61**, 167-234.

Pest Plants and Animals Act (2005) Australian Capital Territory, ACT Parliamentary Counsel. ACT

Peterson, SH, Moritz, MA, Morais, ME, Dennison, PE, Carlson, JM (2011) Modelling long-term fire regimes of southern California shrublands. *International Journal of Wildland Fire* **20**, 1-16.

Philpot, CW (1970) Influence of mineral content on the pyrolysis of plant materials. *Forest Science* **16**, 461-471.

Pickett, BM, Isackson, C, Wunder, R, Fletcher, TH, Butler, BW, Weise, DR (2009) Flame interactions and burning characteristics of two live leaf samples. *International Journal of Wildland Fire* **18**, 865-874.

Plucinski MP, Anderson WR (2008) Laboratory determination of factors influencing successful point ignition in the litter layer of shrubland vegetation. *International Journal of Wildland Fire* **17**, 628-637.

Possell, M, Bell, TL (2013) The influence of fuel moisture content on the combustion of Eucalyptus foliage. *International Journal of Wildland Fire* **22**, 343-352.

Pyne, SJ (1984) Introduction to wildland fire: fire management in the United States.

Pyne, SJ (1990) Firestick History. *The Journal of American History* **76**, 1132-1141.

Pyšek, P, Jarošík, V, Hulme, PE, Pergl, J, Hejda, M, Schaffner, U, Vilà, M (2012) A global assessment of invasive plant impacts on resident species, communities and ecosystems: the interaction of impact measures, invading species' traits and environment. *Global Change Biology* **18**, 1725-1737.

Pyšek, P, Jarošík, V, Pergl, J, Randall, R, Chytrý, M, Kühn, I, Tichý, L, Danihelka, J, Chrtek jun, J, Sádlo, J (2009) The global invasion success of Central European plants is related to distribution characteristics in their native range and species traits. *Diversity and Distributions* **15**, 891-903.

Raison, RJ, Woods, PV, Khanna, PK (1983) Dynamics of fine fuels in recurrently burnt eucalypt forests. *Australian Forestry* **46**, 294-302.

Rambal, S (2001) Hierarchy and Productivity of Mediterranean-Typr Ecosystems. In: Terrestrial Global Productivity. J Roy, S Bernard, HA Mooney. Academic Press: San Diego, California. 575.

Rawson, RP, Billing, PR, Duncan, SF (1983) The 1982–83 forest fires in Victoria. *Australian Forestry* **46**, 163-172.

Red Hill Bush Regeneration Group (2014) Red Hill: A Yellow box/Red Gum grassy Woodland. Retrieved 03, July, 2014, from http://www.redhillregenerators.org.au/red_hill/grassy%20woodland.htm

Rew, LJ, Johnson, MP (2010) Reviewing the Role of Wildfire on the Occurrence and Spread of Invasive Plant Species in Wildland Areas of the Intermountain Western United States. *Invasive Plant Science and Management* **3**, 347-364.

Riaño, D, Chuvieco, E, Salas, J, Palacios-Orueta, A, Bastarrika, A (2002) Generation of fuel type maps from Landsat TM images and ancillary data in Mediterranean ecosystems. *Canadian Journal of Forest Research* **32**, 1301-1315.

Richards, BN, 1990. Biological conservation of the south-east forests: report of the Joint Scientific Committee. Australian Government Public. Service.

Richardson, DM, Gaertner, M (2013) Plant invasions as builders and shapers of novel ecosystems. *Novel ecosystems: intervening in the new ecological world order.* 102.

Richardson, DM, Pysek, P, Rejmanek, M, Barbour, MG, Panetta, FD, West, CJ (2000) Naturalization and Invasion of Alien Plants: Concepts and Definitions. *Diversity and Distributions* **6**, 93-107.

Rinaldi, LMR (2000) Germination of seeds of olive (Olea europaea L.) and ethylene production: effects of harvesting time and thidiazuron treatment. *Journal of horticultural science and biotechnology* **75**, 727-732.

Rinkes, Z, DeForest, J, Grandy, AS, Moorhead, D, Weintraub, M (2014) Interactions between leaf litter quality, particle size, and microbial community during the earliest stage of decay. *Biogeochemistry* **117**, 153-168.

Rodríguez Pleguezuelo, CR, Durán Zuazo, VH, Muriel Fernández, JL, Martín Peinado, FJ, Franco Tarifa, D (2009) Litter decomposition and nitrogen release in a sloping Mediterranean subtropical agroecosystem on the coast of Granada (SE, Spain): Effects of floristic and topographic alteration on the slope. *Agriculture, Ecosystems & Environment* **134**, 79-88.

Rose, S, Fairweather, PG (1997) Changes in floristic composition of urban bushland invaded by Pittosporum undulatum in northern Sydney, Australia. *Australian Journal of Botany* **45**, 123-149.

Rossiter, NA, Setterfield, SA, Douglas, MM, Hutley, LB (2003) Testing the grass-fire cycle: alien grass invasion in the tropical savannas of northern Australia. *Diversity and Distributions* **9**, 169-176.

Rothermel, RC (1972) A mathematical model for predicting fire spread in wildland fuels. USDA Forest Service Technical Report. 41.

Rothermel, RC (1983) How to predict the spread and intensity of forest and range fires. USDA Forest Service Technical Report. 164.

Rothermel, RC (1991) Predicting behavior and size of crown fires in the Northern Rocky Mountains. USDA Forest Service, Intermountain Research Station, Research Paper INT-438. 46.

Rothermel, RC, Deeming, JE (1980) Measuring and interpreting fire behavior for correlation with fire effects. USDA Forest Service Technical Report. 3.

Rowe, N, Speck, T (2005) Plant growth forms: an ecological and evolutionary perspective. *New Phytologist* **166**, 61-72.

Sandberg, DV, Ottmar, RD, Cushon, GH (2001) Characterizing fuels in the 21st century. *International Journal of Wildland Fire* **10**, 381-387.

Saura-Mas, S, Paula, S, Pausas, JG, Lloret, F (2010) Fuel loading and flammability in the Mediterranean Basin woody species with different post-fire regenerative strategies. *International Journal of Wildland Fire* **19**, 783-794.

Scarff, FR, Westoby, M (2006) Leaf litter flammability in some semi-arid Australian woodlands. *Functional Ecology* **20**, 745-752.

Scarff, FR, Westoby, M (2008) The influence of tissue phosphate on plant flammability: A kinetic study. *Polymer Degradation and Stability* **93**,1930-1934.

Schemel, CF, Simeoni, A, Biteau, H, Rivera, JD, Torero, JL (2008) A calorimetric study of wildland fuels. *Experimental Thermal and Fluid Science* **32**, 1381-1389.

Schwilk, DW (2003) Flammability Is a Niche Construction Trait: Canopy Architecture Affects Fire Intensity. *The American Naturalist* **162**, 725-733.

Schwilk, DW, Ackerly, DD (2001) Flammability and serotiny as strategies: correlated evolution in pines. *Oikos* **94**, 326-336.

Schwilk, DW, Caprio, AC (2011) Scaling from leaf traits to fire behaviour: community composition predicts fire severity in a temperate forest. *Journal of Ecology* **99**, 970-980.

Scott, AC (2000) The Pre-Quaternary history of fire. *Palaeogeography, Palaeoclimatology, Palaeoecology* **164**, 281-329.

Scott, JH, Burgan, RE (2005) Standard fire behavior fuel models: a comprehensive set for use with Rothermel's surface fire spread model. *The Bark Beetles, Fuels, and Fire Bibliography*, 66.

Séro-Guillaume, O, Margerit, J (2002) Modelling forest fires. Part I: a complete set of equations derived by extended irreversible thermodynamics. *International Journal of Heat and Mass Transfer* **45**, 1705-1722.

Simpson, RR (2005) Edaphic and vegetation factors controlling carbon and nitrogen pool sizes and greenhouse gas emissions in south-eastern (sub-alpine) Australia. The University of Melbourne. PhD thesis.

Sinden, J, Jones, R, Susie Hester, Odom, D, Kalisch, C, James, R, Cacho, O (2004) The economic impact of weeds in Australia. CRC for Australian Weed Management. University of Adelaide, Adelaide. 56.

Singh, G, Geissler, EA (1985) Late Cainozoic history of vegetation, fire, lake levels and climate, at Lake George, New South Wales, Australia. *Philosophical Transactions of the Royal Society of London. Series B, Biological Sciences* **311**, 379-447.

Singh, JS, Gupta, SR (1977) Plant decomposition and soil respiration in terrestrial ecosystems. *The Botanical Review* **43**, 449-528.

Singh, K, Trivedi, P, Singh, G, Singh, B, Patra, DD (2014) Effect of different leaf litters on carbon, nitrogen and microbial activities of sodic soils. *Land Degradation & Development*.

Singh, P, Upadhayaya, SD (2000) Biological interaction in Tropical Grassland Ecosystems. In: Structure and function in agroecosystem desing and management. M Shiyomi, H Koizumi. CRC Press: Florida, USA. 114-139.

Smith, F (1993) Acacia baileyana, an invasive native: what prevented its earlier spread? Australian National University. Bachelor Honors Dissertation.

Smith, MA, Grant, CD, Loneragan, WA, Koch, JM (2004) Fire management implications of fuel loads and vegetation structure in jarrah forest restoration on bauxite mines in Western Australia. *Forest Ecology and Management* **187**, 247-266.

Sneeuwjagt, RJ, Peet, GB, Beggs, BJ (1979). Forest fire behaviour tables for Western Australia. Forests Department, Western Australia.

Stephens SL, Weise DR, Fry DL, Keiffer RJ, Dawson J, Koo E, Potts J, Pagni PJ (2008) Measuring the rate of spread of chaparral prescribed fires in northern California. *Fire ecology* **4**, 74-86.

Stocker, TF, Qin, D, Plattner, G-K, Tignor, M, Allen, SK, Boschung, J, Nauels, A, Xia, Y, Bex, V, Midgley, PM (2013) Climate change 2013: The physical science basis. Intergovernmental Panel on Climate Change, Working Group I Contribution to the IPCC Fifth Assessment Report (AR5). Cambridge Univ Press, New York.

Stohlgren, TJ, Schnase, JL (2006) Risk Analysis for Biological Hazards: What We Need to Know about Invasive Species. *Risk Analysis* **26**, 163-173.

Stone, LM, Byrne, M, Virtue, JG (2008) An environmental weed risk assessment model for Australian forage improvement programs. *Australian Journal of Experimental Agriculture* **48**, 568-574.

Streeks, TJ, Owens, MK, Whisenant, SG (2005) Examining fire behavior in mesquite–acacia shrublands. *International Journal of Wildland Fire* **14**, 131-140.

Sullivan, AL (2009a) Wildland surface fire spread modelling, 1990–2007. 2: Empirical and quasi-empirical models. *International Journal of Wildland Fire* **18**, 369-386.

Sullivan, AL (2009b) Wildland surface fire spread modelling, 1990–2007. 1: Physical and quasi-physical models. *International Journal of Wildland Fire* **18**, 349-368.

Sullivan, AL (2013) Grassland fire behaviour. CSIRO Bushfire behaviour science symposium. CSIRO. Canberra, ACT. 150.

Sullivan, AL, McCaw, LW, Cruz, MG, Matthews, S, Ellis, PF (2012) Fuel, fire weather and fire behaviour in Australian ecosystems. In: Flammable Australia: fire regimes, biodiversity and ecossistems in a changing world. R Bradstock, AM Gill, RJ Williams. CSIRO: Collingwood, Victoria. 51-77.

Susott, RA (1982) Differential scanning calorimetry of forest fuels. *Forest Science* **28**, 839-851.

Terral J-F, Alonso N*, et al.* (2004) Historical biogeography of olive domestication (*Olea europaea* L.) as revealed by geometrical morphometry applied to biological and archaeological material. *Journal of Biogeography* **31**, 63-77.

Thomas, PB, Watson, PJ, Bradstock, RA, Penman, TD, Price, OF (2014) Modelling surface fine fuel dynamics across climate gradients in eucalypt forests of south-eastern Australia. *Ecography* **37,** 827-837.

Thorp, JR, Lynch, R (2000) The determination of weeds of national significance. National weeds strategy. Launceston.

Toberman, H, Chen, C, Lewis, T, Elser, JJ (2014) High-frequency fire alters C: N: P stoichiometry in forest litter. *Global Change Biology* **20**, 2321-2331.

Tolhurst, KG, McCarthy, GJ, Chatto, K, 1996. Estimating overall fuel hazard for forest fuels. Melbourne, Victoria.

Turnbull, JW (1997) Australian vegetation. In: Australian trees and shrubs: species for land rehabilitation and farm planting in the tropics. Doran, JC, Turnbull, JW. Australian Centre for International Agricultural Research (ACIAR): Australia, Canberra. 19-37.

United Nations Framework Convention on Climate Change (UNFCCC), 2002. A Guide to the Climate Change Convention and its Kyoto Protocol. UNFCCC Secretariat, Bonn. 39.

Van Driesche, RG, Bellows Jr, TS (1996) Biological Control. Chapman & Hall. New York. 539.

Van Nest, TA, Alexander, ME (1999) System for rating fire danger and predicting fire behavior used in Canada. In: National Interagency Fire Behaviour Workshop. Phoenix, Arizona. 1-13.

Van Wagner CE (1973) Height of crown scorch in forest fires. *Canadian Journal of Forest Research* **3**, 373-378.

Van Wagner, CE (1977) Conditions for the start and spread of crown fire. *Canadian Journal of Forest Research* **7**, 23-34.

Van Wilgen, BW, Richardson, DM (1985) The effects of alien shrub invasions on vegetation structure and fire behaviour in South African fynbos shrublands: a simulation study. *Journal of Applied Ecology*. 955-966

Veenendaal, EM, Ernst, WHO, Modise, GS (1996a) Effect of seasonal rainfall pattern on seedling emergence and establishment of grasses in a savanna in south-eastern Botswana. *Journal of Arid Environments* **32**, 305-317.

Veenendaal, EM, Ernst, WHO, Modise, GS (1996b) Reproductive effort and phenology of seed production of savanna grasses with different growth form and life history. *Plant Ecology* **123**, 91-100.

Vigilante, T, Murphy, BP, Bowman, DMJS (2009) Aboriginal fire use in Australian tropical savannas: Ecological effects and management lessons. In: Tropical Fire Ecology: Climate change, land use, and ecosysten dynamics. Cochrane MA. Springer: Chichester, UK. 143-161.

Vilà, M, Espinar, JL, Hejda, M, Hulme, PE, Jarošík, V, Maron, JL, Pergl, J, Schaffner, U, Sun, Y, Pyšek, P (2011) Ecological impacts of invasive alien plants: a meta-analysis of their effects on species, communities and ecosystems. *Ecology Letters* **14**, 702-708.

Vitousek, PM (1986) Biological invasions and ecosystem properties: can species make a difference? In: Ecology of biological invasions of North America and Hawaii. Springer. 163-176.

Vitousek, PM (1990) Biological invasions and ecosystem processes: towards an integration of population biology and ecosystem studies. *Oikos*, 7-13.

Vitousek, PM, D'Antonio, CM, Loope, LL, Rejmanek, M, Westbrooks, R (1997) Introduced species: a significant component of human-caused global change. *New Zealand Journal of Ecology* **21**, 1-16.

Vitousek, PM, Walker, LR (1989) Biological invasion by Myrica faya in Hawai'i: plant demography, nitrogen fixation, ecosystem effects. *Ecological Monographs* **59**, 247-265.

von Richter L, Little D, Benson D (2005) Effects of low intensity fire on the resprouting of the weed African Olive (Olea europaea subsp. cuspidata) in Cumberland Plain Woodland, western Sydney. *Ecological Management & Restoration* **6**, 230-233.

Wagner, CEV (1973) Height of Crown Scorch in Forest Fires. *Canadian Journal of Forest Research* **3**, 373-378.

Walker, J (1981) Fuel dynamics in Australian vegetation. *Fire and the Australian biota* 101-127.

Walther, G-R, Roques, A, Hulme, PE, Sykes, MT, Pysek, P, Kühn, I, Zobel, M, Bacher, S, Botta-Dukát, Z, Bugmann, H, Czúcz, B, Dauber, J, Hickler, T, Jarosík, V, Kenis, M, Klotz, S, Minchin, D, Moora, M, Nentwig, W, Ott, J, Panov, VE, Reineking, B, Robinet, C, Semenchenko, V, Solarz, W, Thuiller, W, Vilà, M, Vohland, K, Settele, J (2009) Alien species in a warmer world: risks and opportunities. *Trends in Ecology & Evolution* **24**, 686-693.

Ward, DJ, Lamont, BB, Burrows, CL (2001) Grasstrees reveal contrasting fire regimes in eucalypt forest before and after European settlement of southwestern Australia. *Forest Ecology and Management* **150**, 323-329.

Watson PJ, Bradstock RA, Morris, EC (2009) Fire frequency influences composition and structure of the shrub layer in an Australian subcoastal temperate grassy woodland. *Austral Ecology* **34**, 218-232.

Watson PJ, Penman SH, Bradstock RA (2012) A comparison of bushfire fuel hazard assessors and assessment methods in dry sclerophyll forest near Sydney, Australia. *International Journal of Wildland Fire* **21**, 755-763.

Watson PJ, Wardell-Johnson G (2004) Fire frequency and time-since-fire effects on the open-forest and woodland flora of Girraween National Park, south-east Queensland, Australia. *Austral Ecology* **29**, 225-236.

Watson, L (1990) The grass family, Poaceae. In: Reproductive versatility in the grass. GP Chapman. Cambridge Press: Melbourne. 1- 31.

Weber, RO (1991) Modelling fire spread through fuel beds. *Progress in Energy and Combustion Science* **17**, 67-82.

Weise, DR, Wright, CS (2014) Wildland fire emissions, carbon and climate: Characterizing wildland fuels. *Forest Ecology and Management* **317**, 26-40.

Wells, G (2008) The Rothermel Fire-Spread Model: still running like a champ. *Fire Science Digest* **2**, 1-12.

Westoby, M (1998) A leaf-height-seed (LHS) plant ecology strategy scheme. *Plant and Soil* **199**, 213-227.

Whelan, R (1995) The ecology of fire. Ed. CU Press. Cambridge University Press: New York. 346.

White, BLA, Ribeiro, AS, Ribeiro, GT, Souza, RM (2013a) Building fuel models and simulating their surface fire bahavior in the "Serra de Itabaiana" national Park, Sergipe, Brazil. *FLORESTA* **43**, 27-38.

White, BLA, Ribeiro, GT, Souza, RM (2013b) O uso do BehavePlus como ferramenta para modelagem do comportamento e efeito do fogo. *Pesquisa Florestal Brasileira* **33**, 73-83.

White, RH, Zipperer, WC (2010) Testing and classification of individual plants for fire behaviour: plant selection for the wildland–urban interface. *International Journal of Wildland Fire* **19**, 213-227.

Williams, D, Baruch, Z (2000) African Grass Invasion in the Americas: Ecosystem Consequences and the Role of Ecophysiology. *Biological Invasions* **2**, 123-140.

Williams, JA, West, CJ (2000) Environmental weeds in Australia and New Zealand: issues and approaches to management. *Austral Ecology* **25**, 425-444.

Williams, RJ, Gill, AM, Moore, PHR (1998) Seasonal changes in fire behaviour in a tropical savanna in northern Australia. *International Journal of Wildland Fire* **8**, 227-239.

Wilson, AAG (1988) Width of firebreak that is necessary to stop grass fires: some field experiments. *Canadian Journal of Forest Research* **18**, 682-687.

Wilson, AAG (1992) Eucalypt Bark Hazard Guide. Victorian Government Publication, Melbourne, Victoria.

Witkowski, ETF (1991) Effects of invasive alien acacias on nutrient cycling in the coastal lowlands of the Cape fynbos. *Journal of Applied Ecology* 1-15.

Wright, IJ, Ackerly, DD, Bongers, F, Harms, KE, Ibarra-Manriquez, G, Martinez-Ramos, M, Mazer, SJ, Muller-Landau, HC, Paz, H, Pitman, NCA, Poorter, L, Silman, MR, Vriesendorp, CF, Webb, CO, Westoby, M, Wright, SJ (2007) Relationships Among Ecologically Important Dimensions of Plant Trait Variation in Seven Neotropical Forests. *Annals of Botany* **99**, 1003-1015.

Wright, IJ, Reich, PB, Westoby, M, Ackerly, DD, Baruch, Z, Bongers, F, Cavender-Bares, J, Chapin, T, Cornelissen, JHC, Diemer, M (2004) The worldwide leaf economics spectrum. *Nature* **428**, 821-827.

Zhang, D, Hui, D, Luo, Y, Zhou, G (2008) Rates of litter decomposition in terrestrial ecosystems: global patterns and controlling factors. *Journal of Plant Ecology* **1**, 85-93.

Zhou, X, Mahalingam, S, Weise, D (2005) Modeling of marginal burning state of fire spread in live chaparral shrub fuel bed. *Combustion and Flame* **143**, 183-198.

Zylstra, PJ (2011) Rethinking the fuel - fire relashionship. In: Bushfire CRC & AFAC 2010 Conference Science Day. Sydney, Australia. RP Thornton. 87-99.