Patterns in plant flammability at a fire-prone wildland-urban interface in eastern Australia

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Certificate of Original Authorship

I, Thomas Hawthorne declare that this thesis, is submitted in fulfilment of the requirements for the award of Master of Science (Research), in the School of Environment at the University of Technology Sydney.

This thesis is wholly my own work unless otherwise referenced or acknowledged. In addition, I certify that all information sources and literature used are indicated in the thesis.

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Statement of thesis format

This thesis is submitted as a thesis by compilation. Chapter 1 provides a general introduction to wildfires and the wildland-urban interface, the gaps in the field and my research questions. The following three data chapters (Chapters 2, 3 and 4) have been prepared as standalone chapters with an additional introductory paragraph to link chapters. Chapter 5 provides a synthesis of my research and highlights the contribution this thesis makes to our understanding of plant flammability in wildland-urban interface systems and provides future directions for the field.

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Abstract

Wildfires can have detrimental impacts on biodiversity, human lives and infrastructure, particularly at the wildland-urban interface (WUI). The WUI consists of low-level urban development nested within a broader area of predominantly natural wildland vegetation, and as such, the WUI is the frontline of human-wildfire interactions. With many WUIs expanding and wildfires becoming more frequent it is critical that we understand wildfire risks and hazards at the WUI for effective future wildfire management. In this context, I investigate patterns in flammability of locally-abundant plant species from urban and wildland areas in a fire-prone WUI in the mid-lower Blue Mountains of eastern Australia.

I show that floristic composition of dominant plant species varies significantly among dry sclerophyll woodland and wet sclerophyll gully forest (wildland) and home gardens and parks and recreational (urban) areas of the WUI. Notably, urban areas have higher exotic species richness and canopy cover which is driven by the prevalence of exotics in household gardens. Given the significant floristic differences between urban and wildland areas, I then compare patterns in shoot flammability across 45 woody plant species between these WUI areas. I show that wildland plants are more flammable than urban plants via their longer burn times (higher sustainability) and larger amounts of consumed shoot biomass (higher consumability). In terms of native and exotic plants in wildland and urban areas, not only are wildland native plants more flammable than urban exotic plants, but urban native plants also have longer burn times, consume more biomass and in addition reach higher combustion temperatures (higher combustibility) than urban exotic plants. Across all species, I show that in general high-flammability plants contain lower shoot water content and have heavier shoot dry biomass than low-flammability plants. Shoot bulk density and degree of ramification do not explain cross-species variation in flammability, and furthermore, none of the shoot traits measured were significant in explaining differences in shoot flammability among wildland-native, urban-native and urban-exotic species.

This thesis takes a novel approach to understanding fire risk at the WUI through the lens of shoot flammability. My findings enhance our knowledge of how plants burn and provide insights for selecting low flammability plants that may mitigate wildfire impacts on human lives and infrastructure. Furthermore, my work underscores the importance in understanding the influence of plant traits on flammability in fire-prone landscapes.

Chapter 1 | Introduction

1.1 The Global Wildfire Problem

Wildfires are unplanned landscape fires that can sweep across large areas quickly and unpredictably (Moritz *et al.* 2014). Wildfires have been a source of ecological disturbance for the last 100 million years and are widespread across many biomes globally (He *et al.* 2019). Fire is an important source of ecological disturbance that yields healthy ecosystem functioning (Bond and Keeley 2005). Substantial research on wildfires and the evolution of plant traits suggest that fire has been a major ecological disturbance that has shaped plant communities worldwide (Kelly and Brotons 2017; Pausas *et al.* 2017; He *et al.* 2019). However, high intensity wildfires adjacent to urban and suburban areas pose serious risks to wildlife, agricultural land, human lives and infrastructure. Under future climate change projections, wildfires are predicted to become more frequent and more intense globally as a result of rising global temperatures and increased drought periods (Scholze *et al.* 2006; Krawchuk *et al.* 2009). These trends have been observed as global temperatures have increased by approximately 0.2 °C per decade over the past 3 decades, which has led to more severe and widespread drought (Jolly *et al.* 2015). Similarly, global estimates of fire season length showed an increase of up to 18.7% between 1979-2013 (Jolly *et al.* 2015).

The frequency of large uncontrolled wildfires has increased globally posing heightened risks to biodiversity, human lives and infrastructure. Wildfires in recent history have been so large and intense that the term “mega-fires” has been coined to describe them (Stephens *et al.* 2014). In August 2007, wildfires spread across the 2700 km^2 of the western Greece mainland destroying 2000 homes and taking 67 lives (Lainas *et al.* 2016). The 2017 California wildfires destroyed in excess of 10 000 structures and resulted in nearly 50 fatalities (Nauslar *et al.* 2018). The 2019/2020 mega-fires in Australia burned over 17 million hectares of land with fires raging through all states of Australia including Tasmania (APH 2020). The fires resulted in the destruction of over 3000 homes and the loss of 33 human lives (APH 2020) (APH 2020). Whilst at this stage only predictions are currently available, huge losses to biodiversity are forecasted with approximately 327 of 1800 threatened species in Australia potentially facing extinction (Wintle *et al.* 2020). At time of writing, the United States are experiencing their worst wildfires in 18 years, as intense uncontrollable mega-fires are burning in California and Oregon (BBC 2020). So far these wildfires have destroyed thousands of homes and killed over 30 people (BBC 2020).

1.2 Wildfires in Australia

Fire has a long history with Australian flora where fire regimes have likely led to the evolution of fire tolerant and fire-dependant plant communities (Lamont *et al.* 1991; Enright *et al.* 1996; Crisp *et al.* 2011). In Australia, it is likely fire weather conditions will become more frequent under future climate change projections. Fire weather modelling in Australia typically adheres to McArthur's Forest Fire Danger Index (FFDI) to assess weather conditions and the risk of wildfires (Bradstock *et al.* 2009; Clarke *et al.* 2011; Clarke *et al.* 2013). The FFDI is an exponential function of drought, temperature, wind speed and relative humidity (Luke and McArthur 1978). It's predicted that FFDI values will increase across most of Australia, leading to an increase in the number of high, very high, and extreme days of wildfire risk (Clarke and Evans 2019). In particular, major increases in fire weather days are predicted for south-east New South Wales (NSW), south-west NSW and Victoria, with the largest increases predicted for areas where catastrophic wildfires have already occurred (Clarke *et al.* 2011). Fine resolution modelling over south-east Australia indicates that high and extremely high FFDI values are predicted to increase by a maximum of 25% and 70% by 2020 and 2050 respectively (Hennessy *et al.* 2005). By 2050, the greater Sydney region may see an increase in large scale wildfire ignition days of up to 84% following an increase in ambient weather and drought as predicted under future climate change scenarios (Bradstock *et al.* 2009). In 2019, Australia had its warmest and driest year on record, with annual mean temperatures above average for nearly all areas of Australia and only 277mm average national rainfall (BOM 2020). These conditions culminated in the largest wildfires recorded in Australia, far exceeding the large 1939 Black Friday bushfires (Nolan *et al.* 2020). Wildfires burned through an estimated 5.4M Ha in NSW alone with an estimated total area burnt of 8.4M Ha across all of Australia (APH 2020). It is likely that we are now experiencing the conditions that were modelled previously to predict an increase fire frequency and intensity here in Australia (Nolan *et al.* 2020).

1.3 Plant Flammability

Plants act as the major fuel for wildfires, yet species' differences in flammability may dictate the distance of spread and severity of wildfires (Schwilk 2015; Zylstra *et al.* 2016; He *et al.* 2019). Plant flammability is recognised as a core functional plant trait with shoot flammability referenced as the preferred flammability descriptor in trait handbooks (Perez-Harguindeguy *et al.* 2013). Leaves make up the bulk of fuels available to wildfires for combustion and differences in leaf traits have been shown to correlate with species' differences in plant flammability. Traits such as leaf mass per area (LMA), leaf water content (LWC), leaf thickness, leaf length, leaf area (LA), leaf volatile organic compounds (VOCs), phosphate concentrations, and the amount of dead material retained by a plant may influence flammability rankings of plants (Behm *et al.* 2004; Blackhall *et al.* 2012; Murray *et al.* 2013; Bianchi and Defosse 2015; Krix and Murray 2018; Dent *et al.* 2019). When measuring the flammability of plant parts, a species' flammability is typically quantified by its ignitability, combustibility, consumability and sustainability components. Ignitability is the time taken for a sample to ignite after exposure to a heat source and relates to the ignition potential of a wildfire (Anderson 1970). Combustibility is the rapidity of the combustion after ignition and is best related to fire intensity (Anderson 1970). Consumability is the proportion of mass or volume of sample that remains after combustion and has implications rates of fire spread (Martin *et al.* 1993). Sustainability is the ability of a sample to sustain combustion once ignited with or without an ignition source which relates to fire residence time and also rates fire of spread (Anderson 1970). Studies will generally choose to address all four components of flammability although this is not always the case (Pellizzaro *et al.* 2007). Whilst some contention exists between defining flammability axes when fire's evolutionary role is discussed (Pausas *et al.* 2017), from a functional and applied ecology standpoint relative flammability of the widely-used axes outlined by Anderson (1970) are sufficient for underpinning interspecific differences in flammability (Murray *et al.* 2020).

Plant flammability has formerly been assessed at the leaf scale with the ignition of leaf samples followed by the measuring of the aforementioned flammability components (Murray *et al.* 2013; Krix and Murray 2018). Leaf scale flammability studies often involve combustion of single leaves in a muffle furnace (Murray *et al.* 2013; Grootemaat *et al.* 2015; Mason *et al.* 2016; Krix and Murray 2018) or an infrared quartz epiradiator (Alessio *et al.* 2008; Blackhall *et al.* 2012; Bianchi and Defosse 2015; Simpson *et al.* 2016). Typically, this procedure led to the measuring of leaf morphological traits in the hope that variation in these traits could explain the large variation in plant flammability (Alessio *et al.* 2008). However, there has been a shift towards analysing plant flammability at the shoot scale using a plant BBQ (Jaureguiberry *et al.* 2011; Burger and Bond 2015; Calitz *et al.* 2015; Wyse *et al.* 2016; Battersby *et al.* 2017a; Battersby *et al.* 2017b; Cubino *et al.* 2018; Wyse *et al.* 2018; Alam *et al.* 2019; Dent *et al.* 2019; Santacruz-Garcia *et al.* 2019). The plant BBQ is a portable device used to quantify the four components of plant flammability (ignitability, sustainability, combustibility and consumability) whilst maintaining the branching architecture of the plant of interest (Jaureguiberry *et al.* 2011; Perez-Harguindeguy *et al.* 2013). Concerns have been expressed that leaf scale laboratory tests may not accurately scale up to whole plant flammability (Fernandes and Cruz 2012). Alam *et al.* (2019) has shown that muffle furnace experiments at the leaf scale do not adequately scale up to whole plant flammability rankings based on expert opinion. Shoot level flammability assessments conducted with the plant BBQ have the potential to accurately predict whole plant flammability (Wyse *et al.* 2016). The procedure for assessing shoot flammability shares similarities with leaf flammability procedures where plant morphological traits are quantified prior to ignition of the sample. At the shoot scale, both leaf and shoot traits are measured including fuel bulk density (BD), fuel moisture content (FMC), dead material retention, degree of ramification (DR), leaf and twig dry matter content (LDMC/TDMC), leaf area (LA) and leaf thickness (Burger and Bond 2015; Wyse *et al.* 2016; Alam *et al.* 2019; Santacruz-Garcia *et al.* 2019). To quantify flammability components at the shoot level, studies typically record time to flame (ignition) as ignitability, burn time (BT) as sustainability, maximum temperature (MT) by the sample as combustibility, and the percentage of biomass consumed (BC) during combustion as consumability (Wyse *et al.* 2016; Cubino *et al.* 2018; Alam *et al.* 2019; Santacruz-Garcia *et al.* 2019). Santacruz-Garcia *et al.* (2019) took a trait-based approach to plant flammability and showed there is potential to rank species' flammability based on their functional traits. Considering shoot flammability literature is growing rapidly and becoming the new standard for measuring plant flammability (Perez-Harguindeguy *et al.* 2013) there remains no shoot scale flammability literature pertaining to the vegetation of Australian ecosystems. Plant flammability research at the shoot scale could provide another lens for informing fire management at the WUI. Many Australian ecosystems are considered fire-prone vegetation assemblages (Murphy et al. 2013). Despite the fire-proneness of Australian vegetation there is a paucity of literature pertaining to vegetation composition at WUIs and even fewer studies that examine the flammability plants at these vegetation junctions. The present research is unique in the way that it takes a plant flammability based approach to examining wildfire risk in urban and wildland vegetation assemblages of the GBMWHA.

1.4 The Wildland-Urban Interface

The wildland-urban interface (WUI) is defined as a spatial system that consists of low-level urban development nested within a broader region of natural wildland vegetation (Michael *et al.* 2018). There are two distinct vegetation types at the WUI: natural vegetation on the wildland side of the interface and urban vegetation on the urban side of the interface. Natural vegetation is representative of unique vegetation assemblages of the particular region at the interface. Urban vegetation consists of plant species present in public parks, sporting ovals and recreational spaces, gardens at shopping centres and carparks, and household gardens. Urban vegetation can include locally native plant species through retention of existing vegetation (Villaseñor *et al.* 2016), yet it often includes exotic ornamental plant species (Etlinger and Beall 2005). WUI systems can have interface WUIs which is where urban area abut wildland vegetation and intermix WUIs where substantial fuel loads exist within low level urban development (Bento-Gonçalves and Vieira 2020).

Understanding wildfire dynamics at the WUI is integral to informing potential mitigation and hazard reduction techniques (Haight *et al.* 2004). There is a large body of literature that has focused on quantifying wildfire risk and ignition probability at the WUI (Haight *et al.* 2004; Zhang *et al.* 2008; Poudyal *et al.* 2012; Calkin *et al.* 2015; Michael *et al.* 2018). However, floristic composition of vegetation assemblages at Australian WUIs is poorly understood. This has implications for wildfire hazards at WUIs as there is enormous variation in plant flammability which is underpinned by species differences in leaf and shoot morphological traits (Murray *et al.* 2013; Krix and Murray 2018). Therefore to better understand wildfire hazards at the WUI we need to better understand floristic composition at WUIs (Zylstra *et al.* 2016).

1.5 Study Region

This research in this thesis is situated in a WUI in Greater Blue Mountains World Heritage Area (GBMWHA) where wildfire has occurred for millions of years (Hammill and Tasker 2010) with large wildfires occurring more recently (Nolan *et al.* 2020). The wildland region of the GBMWHA broadly consists of Eucalypt-dominated vegetation assemblages (Hammill and Tasker 2010). Eucalyptus forest is among the world's most flammable vegetation assemblages (Bond and Keeley 2005; Crisp *et al.* 2011). Often WUI studies have focused on looking at quantifying wildfire risk using geospatial analytics that map WUIs and attempt to understand which areas may be high or low risk based on fire history and distance to vegetation (Haight *et al.* 2004; Price *et al.* 2015; Bento-Gonçalves and Vieira 2020). Whilst this is an important approach to understanding wildfire dynamics, patterns in plant composition and flammability remain undescribed in Australian WUIs. Floristic composition is important in predicting wildfire severity in Australian vegetation assemblages (Zylstra *et al.* 2016).

1.6 Research objectives and thesis outline

In this thesis, I aim to better understand plant flammability at a fire-prone WUI. In Chapter 2, through a community-level approach I elucidate the floristic differences of dominant plant species between WUI vegetation assemblages. Given that plant functional trait variation underpins differences in plant flammability, I will quantify and compare the relative shoot flammability of dominant plants species from WUI plant assemblages in Chapter 3. Finally in Chapter 4, I explore plant functional trait relationships with shoot level flammability axes to better understand which functional traits drive variation in shoot flammability. To fulfil these aims I have investigated the following research questions in Chapters 2-4:

(1) How do floristic composition, species richness, the distribution of plant growth forms and phylogenetic diversity of the dominant plant species differ among the plant communities? (Chapter 2)

(2) What contributions do native and exotic plant species make to observed differences among plant communities in floristic composition, species richness, growth form, distribution and phylogenetic diversity? (Chapter 2)

(3) How do dominant WUI plant species vary in the flammability attributes time to flame, burning time, biomass consumed and combustion temperature? (Chapter 3)

(4) How do native and exotic species from urban and wildland areas differ in the flammability attributes; time to flame, burning time, biomass consumed and combustion temperature? (Chapter 3)

(5) Can shoot traits explain interspecific variation in the flammability attributes of locally abundant WUI plants? (Chapter 4)

(6) Can shoot traits explain differences between native and exotic species from urban wildland areas in this WUI? (Chapter 4)

Chapter 2 | Floristic patterns among the dominant plant species of the wildland-urban interface in the mid-to-lower Blue Mountains of eastern Australia

2.1 Introduction

Plant flammability varies enormously among species (Anderson 1970; Hogenbirk and Sarrazindelay 1995; Alessio *et al.* 2008; Cui *et al.* 2020b). Differences in flammability are driven by interspecific variation in traits that determine how fast plants ignite (ignitibility), how long they burn for (sustainability), the intensity with which a fire burns (combustibility) and how much of the plant is burned by fire (consumability)(Anderson 1970; Martin *et al.* 1993; White and Zipperer 2010; Murray *et al.* 2013; Grootemaat *et al.* 2015; Simpson *et al.* 2016; Wyse *et al.* 2016; Cubino *et al.* 2018; Krix *et al.* 2019; Santacruz-Garcia *et al.* 2019). Species level differences in flammability, along with floristic variation in ecological communities across landscapes (Whittaker 1965; Murray *et al.* 1999; Souza and Eisenlohr 2020) means that we are likely to observe variation in vegetation flammability across communities. Indeed, recent empirical studies have shown that shifts in floristic patterns among communities do generate community-level differences in flammability (Cubino *et al.* 2018; Krix and Murray 2018), and differences in floristics have been shown to influence fire behaviour in experimental burns (Cardoso *et al.* 2018; Wragg *et al.* 2018). Thus, determining the floristic composition of vegetation assemblages is a critical first step to determine and compare inherent wildfire hazards of different plant communities in an area. One part of the landscape where wildfire hazard can be high is the wildland-urban interface (WUI).

The wildland-urban interface describes areas where low-level urban development is nested within a broader area of predominantly natural wildland vegetation (Michael *et al.* 2018; Bento-Gonçalves and Vieira 2020). Assessing the wildfire hazard of common plants within these urban areas and in surrounding wildland vegetation is paramount, as the WUI is the frontline of human-wildfire interactions. Thus, there is a heightened risk of loss of human lives and infrastructure in these areas (Gill *et al.* 2013; Calkin *et al.* 2014; Moritz *et al.* 2014; Penman *et al.* 2015). Understanding wildfire hazards at the WUI is integral to informing potential mitigation and hazard-reduction techniques (Haight *et al.* 2004). Floristically, wildland areas at the WUI typically contain native plant species characteristic of the particular region (Lampin-Maillet *et al.* 2010; Villaseñor *et al.* 2016) but can contain plantation forests and agricultural land (Bento-Gonçalves and Vieira 2020). In contrast, urban areas typically include exotic plant species, a pattern driven by the horticultural industry that promotes the inclusion of exotic plants, particularly in household and public gardens (Daniels and Kirkpatrick 2006; Goddard *et al.* 2010; Lerman and Warren 2011). In addition, both urban and wildland plant communities may have received accidental exotic plant invaders as a result of global trade pathways (Hulme 2009).

In this chapter, I performed floristic surveys of the dominant plant species in wildland and urban areas of the WUI in the mid-to-lower Blue Mountains in the Greater Blue Mountains World Heritage Area (GBMWHA) in eastern Australia. In this study region, wildland plants in proximity to urban areas belong to predominantly one of two vegetation associations, either dry sclerophyll woodland or wet sclerophyll gully (Hammill and Tasker 2010). Urban plants occur mainly in one of two types of locations, either in household gardens or in parks and recreational areas. I compared these four areas to address the following questions:

(1) How do floristic composition, species richness, the distribution of plant growth forms and phylogenetic diversity of the dominant plant species differ among the WUI areas?
(2) What contributions do native and exotic plant species make to observed differences among WUI areas in floristic composition, species richness, growth form distribution and phylogenetic diversity?

This study targets the dominant species of plant assemblages as their plant canopy coverage occupies the most physical space in assemblages. As such, these dominant plant species are likely to have a relatively strong influence on wildfire dynamics at the WUI. The answers to the two questions posed above will therefore provide important baseline information for assessing the relative wildfire risks of the four key areas at the WUI. In the following chapter, I perform flammability experiments using a selection of the species identified in this chapter to assess relative differences in flammability between common urban and wildland plants from the four key areas at the WUI.

2.2 Methods

2.2.1 Study region and sites

The mid-to-lower Blue Mountains region is approximately 54 km west of Sydney on the south-eastern coast of Australia (Fig. 2.1). The Greater Blue Mountains World Heritage Area possesses globally unique natural values and is heritage listed because of its ecologically significant flora and fauna, natural habitats for the conservation of biodiversity and threatened species of outstanding value (Hammill and Tasker 2010). The Blue Mountains is a temperate region with warm summers (mean maximum temperature 29°C) and cool winters (mean annual temperature 16°C), and a mean annual rainfall of 1399.6 mm (BOM 2020). The study region spanned the lower-eastern end of the Blue Mountains from the town of Lapstone to the mid-mountain town of Faulconbridge (Fig. 2.2).

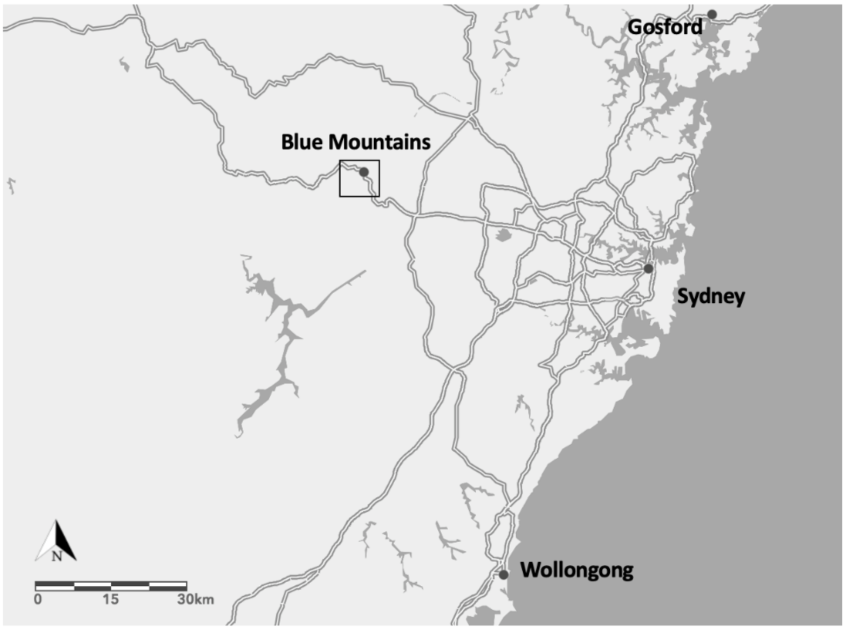

Fig. 2.1 The proximity of the Blue Mountains to other major cities of the central coast and hinterland of NSW. The area in the small box encapsulates the study region and is expanded below in Fig. 2.2.

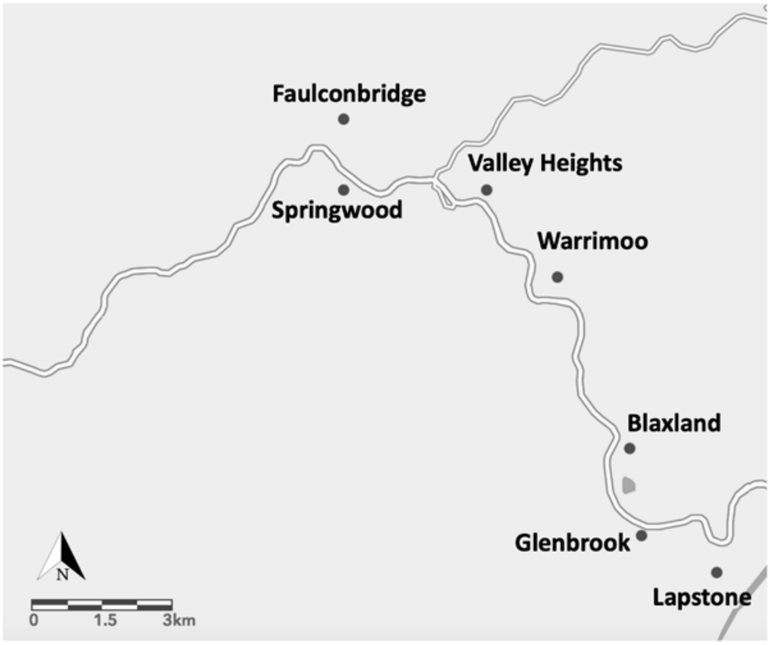

Fig. 2.2 Suburbs of the study region across an elevational gradient from Lapstone (160 m a.s.l.) to Faulconbridge (446 m a.s.l.).

The elevational gradient covered a total of seven suburbs with Lapstone (33° 46' 26.4'' N, 150° 38' 13.2'' E) the lowest elevation suburb at 160 m.a.s.l and Faulconbridge (33° 41' 10.752'' S, 150° 33' 0.504'' E) the highest elevation suburb at 446 m.a.s.l (Fig. 2.2). These seven suburbs represent urban habitats nested within broad areas of wildland vegetation.

To select study sites, I initially located 20 prospective sites in each suburb on a topographical map of the study region. These 20 sites consisted of five sites in each of dry sclerophyll woodland (DS), wet sclerophyll gully (GU), household gardens (HG) and public parks and recreational grounds (PR). Sites were 100 m^2 in area and were selected so that they were separated by a minimum distance of 100 m. Sites in both GU and DS were situated on road edges adjacent to wildland vegetation and in close proximity to urban development so as to capture those wildland species at the immediate interface between wildland and urban areas. After visiting the 140 prospective sites, the vegetation at 110 sites were surveyed across the seven suburbs of the study region. The 30 sites omitted from further consideration were either not accessible as they were on private land or had experienced severe disturbance (such as land clearing) that meant that they were not representative of intact plant communities. I surveyed 56 sites representative of wildland plant communities consisting of 24 GU sites and 32 DS sites. I surveyed 54 sites representative of urban plant communities consisting of 35 HG sites and 19 PR sites.

2.2.2 Vegetation surveys

Vegetation surveys began on the 15 May 2018 and were completed by late August 2018. Wildland areas were surveyed using a 20 m long by 5 m wide quadrat at each site. Quadrats were placed parallel to road edges with the 5 m width running into vegetation and perpendicularly to the road to capture a diverse range of plant species directly at the interface of wildland and urban areas. At urban sites, quadrat area was kept consistent with wildland sites, however, dimensions were altered where necessary to fit the spatial arrangement of household gardens (e.g. 10 m x 10 m or 12.5 m x 8 m were most appropriate). Within each sampling quadrat, I recorded the identities and local abundances (percentage projected canopy cover) of all dominant plant species (i.e. > 25% canopy cover) to assess floristic composition. Local abundance was measured by visually assigning each plant species within a quadrat to one of three canopy cover classes following the Braun-Blanquet method (Sutherland 2006): class 1 (25–50% cover), class 2 (51–75% cover), class 3 (76–100% cover). Plant species richness at each site was measured as a count of the number of species identified within a quadrat. The plant stratum in which each species occurred was recorded at each site as either understorey (0–1 m), midstorey (1–3 m) or overstorey (> 3 m). Species present in more than one stratum were recorded as present in all strata in which they occurred as they contributed potentially flammable foliage in the different strata. Plant identification to species level (and below) was based on the nomenclature of Robinson (1991) and Fairley and Moore (2010).

2.2.3 Data analyses

I employed redundancy analysis (RDA) to compare floristic composition among the four WUI areas. Abundances (canopy cover classes) of all species within each site were used in the analysis and all sites were included as replicates with suburb entered as a conditioning factor. Prior to the RDA the site x species matrix was Hellinger transformed (Legendre and Gallagher 2001). I used permutational ANOVA (9999 permutations) of the RDA to test if there was an overall significant difference in floristic composition among the four WUI areas (four-level fixed explanatory variable: DS, GU, HG, PR). This was followed by a series of pairwise permutational ANOVA comparisons (9999 permutations) to determine which areas differed significantly from each other. I calculated the proportion of sites that each species occupied and each species' mean local abundance (canopy cover class) separately for the four WUI areas to better understand which species most strongly contributed to differences in floristic composition. I graphically compared these values to identify species with the highest occupancy and species with the highest local abundance within each area.

A generalized linear model (GLM) with a Poisson error structure and log link was used to model total (native plus exotic) plant species richness (response variable) as a function of WUI area (four-level fixed explanatory variable: DS, GU, HG, PR) and suburb (seven-level fixed explanatory variable, one level for each suburb), with the inclusion of a WUI area x suburb interaction term. Total plant species richness was modelled separately within each stratum (understorey, midstorey, overstorey) using zero inflated Poisson GLMs with the same model structure described above. Analyses across all strata and for each stratum were then performed separately for the pools of native and exotic plant species. Where a significant difference was found, post-hoc pairwise Z tests were performed to identify significant differences between WUI areas.

The distribution of growth forms of species with cover > 25% in a quadrat across the four WUI areas was investigated using a χ^2 test. A contingency table of counts of herbs, grasses, shrubs and trees by area was generated across all sites, with ferns excluded for having expected cell counts < 5, and following the χ^2 test, patterns in growth form distribution were interpreted using the size and sign of Pearson residuals (Agresti 2019).

To assess differences in phylogenetic diversity among the four WUI areas, a phylogenetic tree of the study species was generated using information on plant phylogenetic relationships in (Zanne *et al.* 2014). Phylogenetic diversity (Faith 1992) was calculated using all species across all sites within each area (DS, GU, PR, HG). Replicates in the analyses described below were calculations of the relative distances along branches for species pairs within each site. Two models – each a quasi-Poisson GLM to account for the distribution of the non-integer data – were fitted to the data. The first used only WUI area as an explanatory factor (four-level fixed explanatory variable: DS, GU, HG, PR) while the second used WUI area, species richness and a WUI area x species richness interaction term. The first model was used to determine which areas differed in phylogenetic diversity. The second model was implemented to determine whether phylogenetic diversity within areas increased with increasing species richness at the same rates within each of the four WUI areas. Post-hoc pairwise Z tests were performed to identify significant differences between areas in the first model and between slopes of the relationships between species richness and phylogenetic diversity in the second model.

All statistical analyses were performed using R (R core team 2019), using the packages *car* (Fox and Weisberg 2018), *emmeans* (Lenth *et al.* 2018), *picante* (Kembel *et al.* 2010), *pscl* (Jackman 2017), *vegan* (Oksanen *et al.* 2019) and *RVAideMemoire* (Hervé 2019).

2.3 Results

2.3.1 Floristic comparisons among plant communities

In total, 102 species (64 native, 48 exotic) were recorded (where canopy cover was > 25% of a quadrat) across the vegetation surveys (Appendix 2.1), with 42 native and one exotic species in DS, 31 native and two exotic species in GU, 12 native and 19 exotic species in HG and 24 native and 9 exotic species in PR. I found a significant difference overall in floristic composition among the four plant communities ($F_{3,103} = 4.32$, $P = 0.001$; Fig. 2.3). Indeed, significant differences emerged in almost all pairwise comparisons between communities ($P < 0.001$ for DS vs GU, DS vs HG, DS vs PR, GU vs HG, HG vs PR), with the only exception being the comparison between GU and PR ($P = 0.50$).

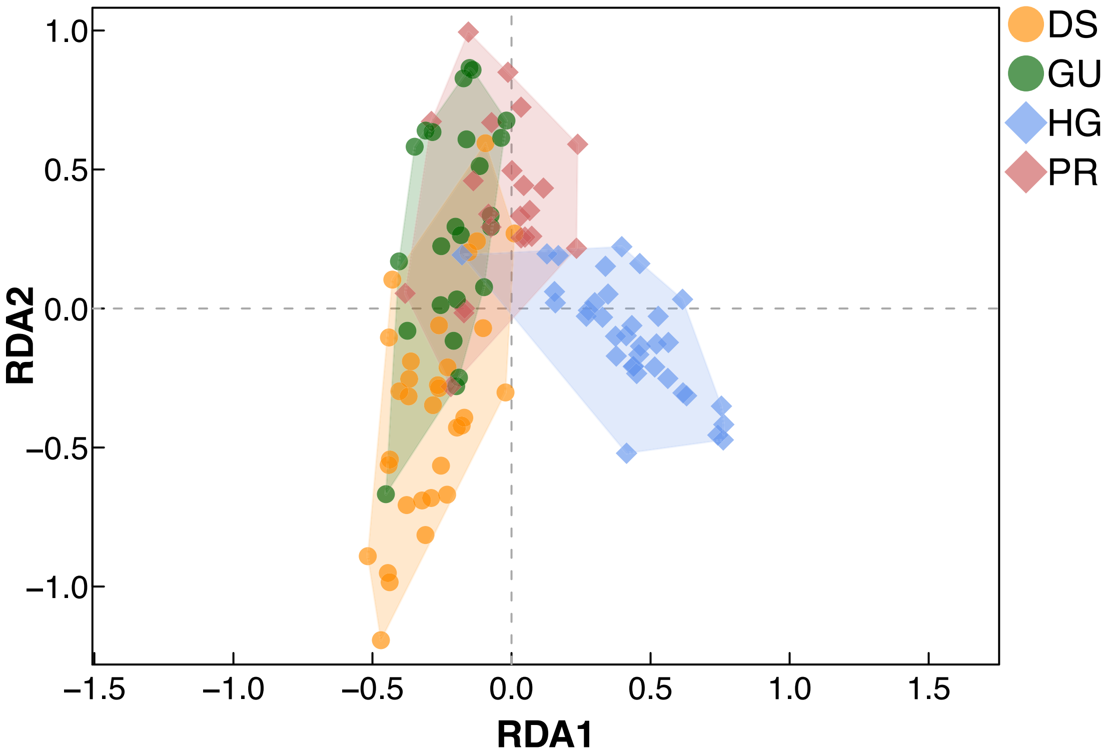

Fig. 2.3 RDA biplot showing site scores with the four WUI areas dry sclerophyll woodland (DS), wet sclerophyll gully (GU), household gardens (HG) and public parks and recreational grounds (PR). Shaded areas of the same colour as the points show the convex hull of the site scores.

The prevalence of exotic species in HG, particularly the widely distributed and highly abundant *Agapanthus praecox* and *Camellia sasanqua*, contributed to HG differing floristically from DS, GU and PR in the study region (Figs 2.4, 2.5). Furthermore, many exotic species were found infrequently but in high local abundance in HG and nowhere else, including *Aechmea cylindrata*, *Camellia japonica*, *Cycas revoluta*, *Nephrolepis exaltata*, *Photinia glabra*, *Pinus radiata*, *Pittosporum tenuifolium*, *Rhododendron degronianum*, *Tropaeolum majus*, *Viburnum odoratissimum* and *Yucca filamentosa* (Fig. 2.5). It is worth noting, however, that some exotic species did occur at cover-abundances of >25% – but only infrequently – in PR (*Asparagus aethiopicus*, *Buxus microphylla*, *Cryptomeria japonica*, *Jacaranda mimosifolia*, *Lavandula angustifolia*, *Liquidambar styraciflua*, *Quercus robur*, *Ulmus minor*, *Ulmus parvifolia*), GU (*Imperata cylindrica*, *Lantana camara*) and in DS (*Asparagus aethiopicus*). Only 12 native species were recorded in HG with none particularly widespread (Fig. 2.4). However, *Angophora costata*, *Callistemon viminalis*, *Corymbia gummifera*, *Lomandra longifolia*, *Melaleuca quinquenervia*, *Patersonia fragilis* and *Pittosporum undulatum* were found in high local abundance where they occurred in HG. Importantly, the most widespread and abundant native species in DS (*Corymbia gummifera*, *Poa labillardierei*, *Syncarpia glomulifera*) and GU (*Calochlaena dubia*, *Pittosporum undulatum*) were recorded infrequently if at all in HG (Figs 2.4, 2.5).

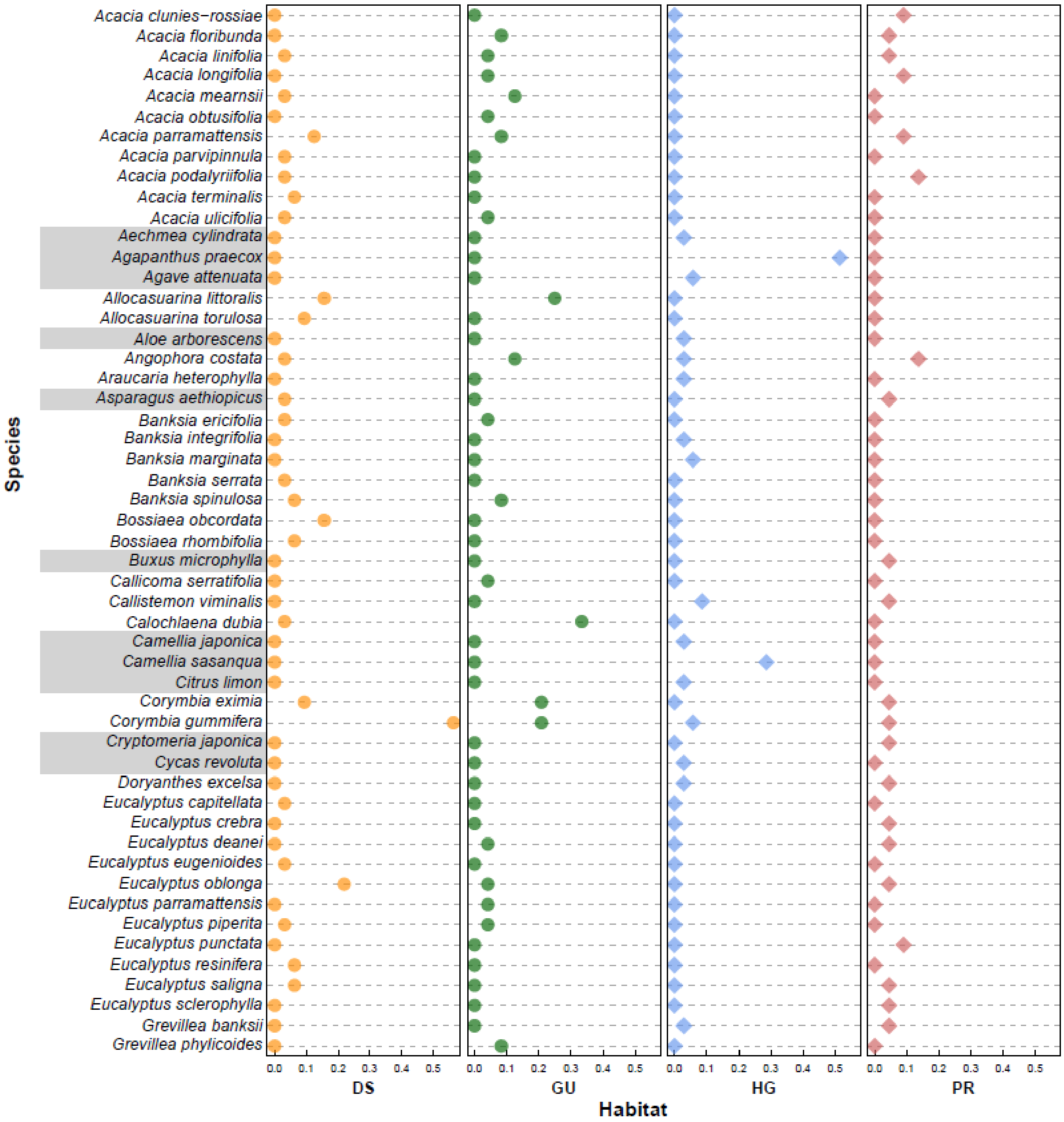

Fig. 2.4 Part 1 The distributions of dominant plant species among dry sclerophyll woodland (DS), wet sclerophyll gully (GU), household gardens (HG), and public parks and recreational (PR) sites and the proportion of sites occupied by each species (i.e. occupancy) within each area. The names of non-native species are shaded in grey.

Fig. 2.4 Part 2 The distributions of dominant plant species among dry sclerophyll woodland (DS), wet sclerophyll gully (GU), household gardens (HG), and public parks and recreational (PR) sites and the proportion of sites occupied by each species (i.e. occupancy) within each community. The names of non-native species are shaded in grey.

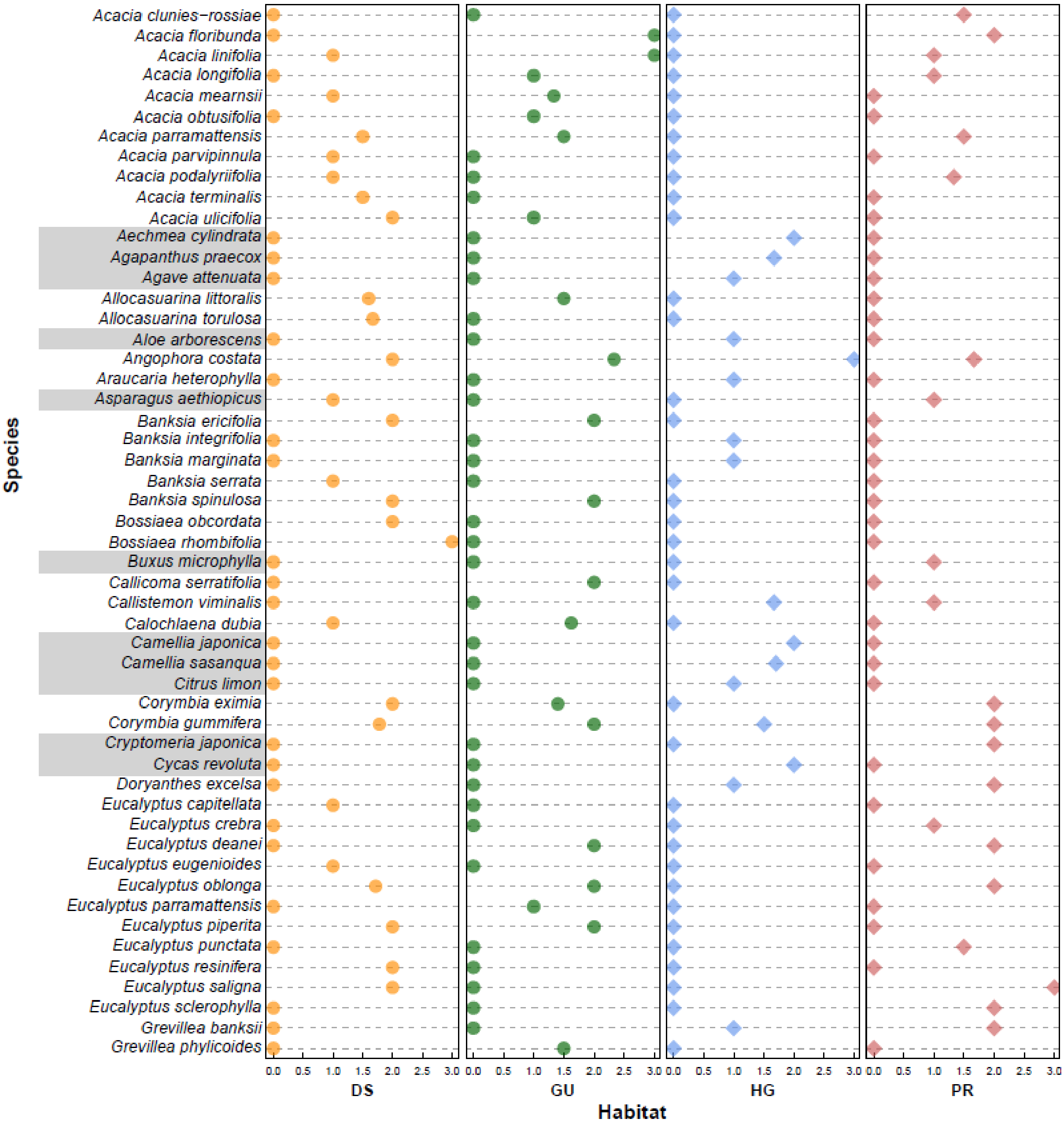

Fig. 2.5 Part 1 Mean local abundances of dominant plant species among dry sclerophyll woodland (DS), wet sclerophyll gully (GU), household gardens (HG), and public parks and recreational (PR) sites. Local abundance was measured as class 1 (25–50% cover), class 2 (51–75% cover), class 3 (76–100% cover). The names of non-native species are shaded in grey.

Fig. 2.5 Part 2 Mean local abundances of dominant plant species among dry sclerophyll woodland (DS), wet sclerophyll gully (GU), household gardens (HG), and parks and recreational (PR) sites. Local abundance was measured as class 1 (25–50% cover), class 2 (51–75% cover), class 3 (76–100% cover). The names of non-native species are shaded in grey.

There was a significant difference in floristics between DS and GU, despite a broad overlap of 22 species in common between the two wildland areas (Figs 2.4, 2.5). A key driver of the difference in floristics was that not only was the fern *Calochlaena dubia* distributed more widely in GU than in DS, but that the tree species *Corymbia gummifera* and *Syncarpia glomulifera* were more widely distributed in DS than in GU (Fig. 2.4). In addition, some species were found infrequently but in high abundance in only one of the two wildland areas. Such species included, for example, *Acacia floribunda*, *Callicoma serratifolia* and *Imperata cylindrica* in GU and *Allocasuarina torulosa*, *Bossiaea obcordata*, *Bossiaea rhombifolia*, *Hakea bakeriana*, *Leptospermum juniperinum*, *Panicum simile*, *Patersonia fragilis*, *Pultenaea linophylla* and *Pultenaea scabra* in DS (Figs 2.4, 2.5). The highly invasive shrub *Lantana camara* was not found in DS, but was found infrequently in GU in high abundance (Fig. 2.5).

Much of the floristic variation between DS and PR was linked to the high abundance of a range of native plant species in DS that were not found at all in PR (Fig. 2.5). This included *Acacia ulicifolia*, *Banksia ericifolia*, *Banksia spinulosa*, *Bossiaea obcordata*, *Bossiaea rhombifolia*, *Hakea bakeriana*, *Leptospermum juniperinum*, *Panicum simile*, *Patersonia fragilis* and *Pultenaea linophylla*. In addition, *Corymbia gummifera* and *Syncarpia glomulifera* were considerably more widespread in DS than in PR (Fig. 2.4). Interestingly, the most widespread species in PR, *Pittosporum undulatum*, was also fairly widespread in DS (Fig. 2.4). Several species were found only in PR and in high abundance, floristically distinguishing PR from DS, including the native species *Acacia floribunda*, *Doryanthes excelsa*, *Grevillea banksii* and *Melaleuca quinquenervia* and the exotic species *Cryptomeria japonica*, *Liquidambar styracifolia* and *Quercus robur* (Fig. 2.5).

2.3.2 Comparisons of species richness among WUI areas

Total richness of dominant plant species differed significantly among WUI areas across all strata (Table 2.1) with significantly lower richness in PR than in DS ($P = 0.04$; Fig. 2.6a). This was underpinned primarily by significantly lower total richness in the understorey (Table 2.1) in PR than in DS ($P = 0.0002$; Fig. 2.6b). Total richness of the understorey was also significantly lower in PR than in HG ($P = 0.0007$; Fig. 2.6b), while total richness in the overstorey (Table 2.1) was significantly lower in HG than in DS ($P < 0.0001$) and GU ($P = 0.047$; Fig. 2.6d). There were no significant differences among WUI areas in total richness of the midstorey (Table 2.1, Fig. 2.6c).

Table 2.1 Tests of differences in total species richness, native and exotic species richness among the four WUI areas, for all plant strata combined and separately for the understorey (US), midstorey (MS) and overstorey (OS). Significant P values in bold.

Species richness	**Plant community** (d.f. = 3)		**Suburb** (d.f. = 6)		**Plant community x suburb** (d.f. = 18)	
	χ^2	P	χ^2	P	χ^2	P
Total species richness						
All strata	8.91	**0.03**	6.83	0.30	7.02	0.99
Understorey	12.22	**0.007**	5.96	0.40	10.93	0.90
Midstorey	6.17	0.10	3.22	0.80	10.02	0.90
Overstorey	15.07	**0.002**	2.94	0.80	10.42	0.90
Native species richness						
All strata	54.36	**< 0.0001**	5.91	0.40	11.29	0.90
Understorey	31.24	**< 0.0001**	6.83	0.30	8.07	0.98
Midstorey	16.54	**0.0009**	1.35	0.97	7.21	0.99
Overstorey	23.34	**0.003**	5.90	0.90	10.80	0.90
Exotic species richness						
All strata	29.33	**0.0003**	5.56	0.90	3.92	1.00
Understorey	21.40	**0.002**	3.30	0.95	1.74	1.00
Midstorey	2.13	0.70	6.33	0.50	0.37	1.00
Overstorey	2.55	0.80	5.64	0.70	0.26	1.00

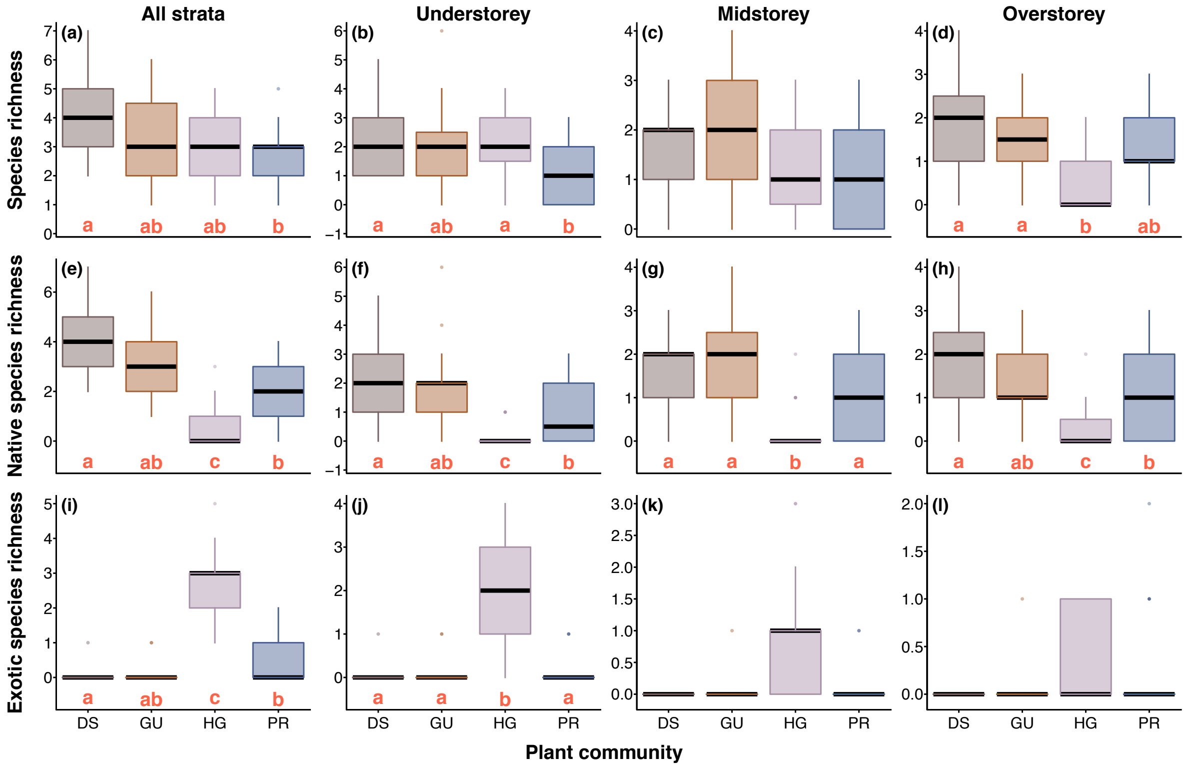

Fig. 2.6 Boxplots of total species richness (plots a-d), native species richness (e-h) and exotic species richness (i-l) for all strata (a, e, i), the understorey (b, f, j), the midstorey (c, g, k) and the overstorey (d, h, l) in the four WUI areas (DS = dry sclerophyll woodland, GU = wet sclerophyll gully, HG = household gardens, PR = public parks and recreational grounds). Red letters below boxplots indicate which areas are significantly different (P < 0.05).

Native plant species richness differed significantly among WUI areas across all strata (Table 2.1) with significantly lower native richness in PR than in DS (P < 0.0001; Fig. 2.6e). This was driven primarily by significantly lower native richness in the understorey (Table 2.1) in PR than in DS (P < 0.0001; Fig. 2.6f), a pattern repeated in the overstorey (P = 0.04; Fig. 2.6h). Across all strata, native richness was significantly lower in HG than in DS (P < 0.0001), GU (P < 0.0001) and PR (P < 0.0001). These significant differences in native richness across the strata were consistent within each stratum (Table 2.1) for HG vs DS (all strata at P < 0.0001), HG vs GU (understorey $P < 0.0001$, midstorey $P < 0.0001$, overstorey $P = 0.001$) and HG vs PR (understorey $P = 0.01$, midstorey $P = 0.002$, overstorey $P = 0.01$; Fig. 2.5f–h).

Exotic plant species richness differed significantly among WUI areas across all strata (Table 2.1) with significantly higher exotic richness in PR than in DS ($P < 0.04$; Fig. 2.6i). In addition, there exotic richness was significantly higher in HG than in DS, GU and PR (all at $P < 0.0001$), which was linked to significantly higher exotic richness in the understorey (Table 2.1) in HG vs DS, HG vs GU and HG vs PR (all at $P < 0.0001$; Fig. 2.6j). There were no significant differences in exotic richness among areas in the midstorey or overstorey (Fig. 2.6k, l).

In all of the analyses described above, there were no significant effects of suburb on species richness and there were no significant WUI area x suburb interactions in any of the models (Table 2.1).

2.3.3 Comparison of growth forms WUI areas

There was a significant difference ($\chi^2 = 102.61$, DF = 9, $P < 0.001$) among the WUI areas in the proportion of species in each growth form (Fig. 2.7). This overall difference was driven primarily by HG having higher proportions of herbs and shrubs in conjunction with lower proportions of grasses and trees compared with the other three areas.

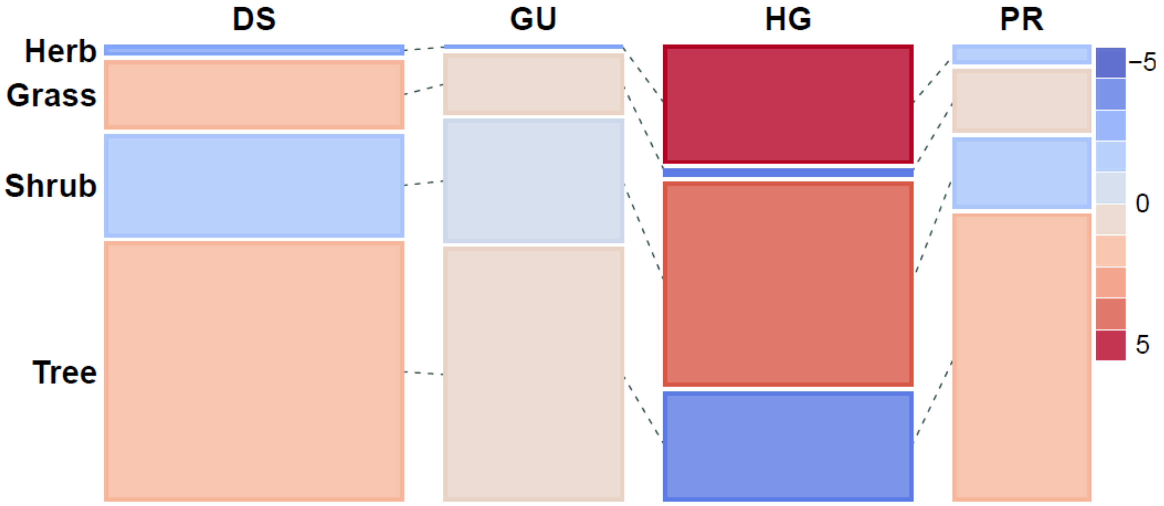

Fig. 2.7 Mosaic plot of the χ^2 contingency table with growth form forming rows, and WUI areas forming columns. Column width indicates the proportion of all study sites in each area. The height of the 'tiles' indicates the proportion of species in each growth form in each area. Tiles are coloured by values of Pearson residuals (key at top right), with values < -3 or > 3 strongly contributing to the significance of the test. Broken lines link tiles for clarity.

2.3.4 Comparison of phylogenetic diversity among WUI areas

Overall, the model of phylogenetic diversity as a function of plant community was significant $\chi^2 = 11.44$, DF = 3, P = 0.01; Fig. 2.8a). There were significant pairwise comparisons between GU and PR (P = 0.008) and between DS and PR (P = 0.03) with higher diversity in both DS and GU than in PR (Fig. 2.8a). Significant effects emerged in the model of phylogenetic diversity as a function of both plant community and species richness for plant community ($\chi^2 = 19.86$, DF = 3, P = 0.0002), species richness ($\chi^2 = 238.441$, DF = 1, P < 0.0001) and the area x richness interaction ($\chi^2 = 9.58$, DF = 3, P = 0.02). The significant interaction arose from the significantly steeper slope for HG relative to DS (P = 0.02), showing that phylogenetic diversity increased more rapidly in HG than in DS as species richness increased in each area.

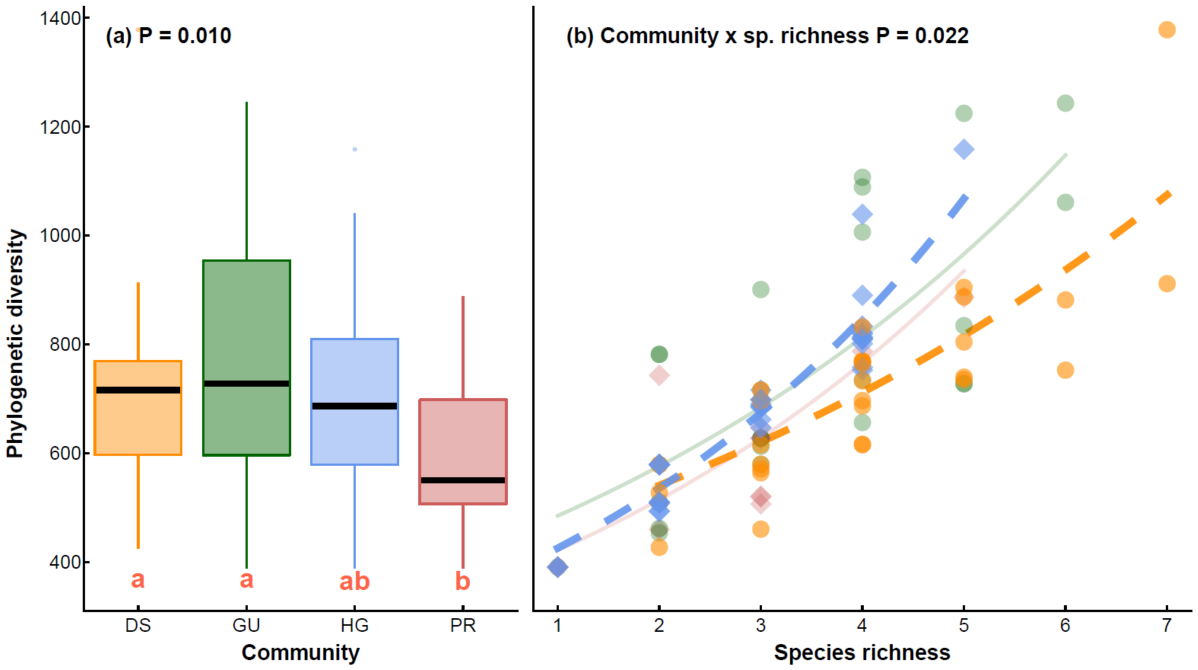

Fig. 2.8 (a) Boxplots of phylogenetic diversity by plant community, with letters below boxplots indicating which communities are significantly different ($P < 0.05$) and (b) coefficients from the plant community x species richness interaction term. Coloured points and lines in (b) match the colours of the plant communities in (a).

2.4 Discussion

The four areas at the WUI of the study region varied significantly in floristic composition, species richness, growth form distribution and phylogenetic diversity. Most notably, the urban area HG differed considerably in floristics from both of the wildland areas DS and GU. This was largely driven by the dominance of exotic plant species in HG. Exotic plant species richness was highest in HG while native species richness lowest. Dominance of exotic species in urban gardens tends to be a consistent pattern in urban areas globally (Tait *et al.* 2005; Smith *et al.* 2006; Acar *et al.* 2007; Dolan *et al.* 2011; Knapp *et al.* 2012; Barrico *et al.* 2018). Exotic species can move into urban areas as a result of deliberate introduction as ornamental species or accidental introduction though global trade (Goddard *et al.* 2010). With globalisation and opening up of international trade pathways, homeowners have a greater choice for which plants they may choose for their gardens (Hulme 2009). Indeed, this is represented in the significantly faster increase in phylogenetic diversity with increasing species richness that we found in HG compared with DS. Plant choice may be dependent on a range of factors but most commonly homeowners and landscapers choose plants of high aesthetic value that are often ‘low maintenance’ (Larson *et al.* 2016). Across all strata total species richness did not differ between urban and wildland vegetation assemblages but overstorey species richness was lower in HG than the other wildland assemblages. This difference in overstorey richness may be a result of homeowners removing trees and other tall shrubs that may present danger to the house during severe weather events.

In the context of bushfire management the most important comparison in this study is between urban vegetation (PR and HG) and wildland vegetation (DS and GU). Understanding how vegetation differs at WUIs is integral to effective bushfire management in WUI areas (Murray *et al.* 2020). Given that plant flammability is highly variable between species (Murray *et al.* 2013; Cui *et al.* 2020b), exotic species have the potential to change flammability dynamics at the WUI. Invasive exotic plants can enter a system and thereby change wildfire regimes through altering fuel structure or fuel composition (Brooks *et al.* 2004). Exotic plants may possess markedly different morphological traits than the native vegetation which can therefore increase or decrease species’ community flammability and alter fire regimes (Gorgone-Barbosa *et al.* 2015). Cubino *et al.* (2018) demonstrated a decline in community level flammability following the long term (25 years) invasion of exotic forbs in tussock grasslands in New Zealand. However, in the Top End of northern Australia, the invasion of the exotic gamba grass (*Andropogon gayanus*) significantly increased community flammability through forming taller, denser fuel loads that could support fires eight times more intense than native species (Rossiter *et al.* 2003). This suggests that exotic invasions can have variable effects and their potential influence on an environment is related to differences in plant morphology and therefore plant flammability. HG sites were by far the greatest source of exotic plant species in this study with the highest exotic species richness and abundance of the four WUI areas. It is important to consider how wildfire risk may change at the WUI with the addition of exotic plants from domestic gardens (Murray *et al.* 2020). Exotic domestic garden plants can have morphological traits that may result in them attaining high flammability rankings. Research on WUI ornamental plants at the leaf scale showed that domestic plants have the potential to be highly flammable (Ganteaume *et al.* 2013). In particular, leaf-scale ignitability of *Photinia x fraseri*, a hybrid of *Photinia glabra* was among the highest of the study into flammability of WUI ornamental species (Ganteaume *et al.* 2013). Similarly, at the shoot scale Cui *et al.* (2020b) found *P. glabra* to be high flammable in terms of ignitability and combustibility. *Photinia glabra* was a WUI species of the present study region as a hedge species in HG sites. Similarly, Murray et al. (2013) showed dry leaf flammability of exotic species was significantly higher than native species in dry sclerophyll woodland. *Olea europea* and *Lantana camara* were among the exotic species tested by Murray *et al.* (2013) and were common to urban vegetation assemblages in the present WUI. Fresh leaf ignitability of exotic leaves in this study were generally slower, driven by high moisture content of exotic leaves compared to relatively low moisture content in native leaves (Murray *et al.* 2013). This was consistent with Berry *et al.* (2011) who found *Lantana camara* to be the least flammable species among dry rainforest species of QLD which was reflected by *L. camara's* low leaf and twig dry matter content. However, *L. camara* invasions significantly increased the woody understorey biomass increasing fire risk in this dry rainforest (Berry *et al.* 2011).

Interestingly, there was no significant difference in floristics between PR and GU areas. These areas may be similar due to the planting of popular cultivated species with high aesthetic value that may be shared between GU and PR sites. It is not uncommon to find a high proportion of native species in public gardens (Daniels and Kirkpatrick 2006; Kendal *et al.* 2012). The graminoid *Lomandra longifolia* was found to have high occupancy and abundance in both GU and PR and is often utilised in landscape design in Australia (CPS 2018; CoS 2020). Similarly, the trees, *Lophostemon confertus* and *Syncarpia glomulifera*, are widely-used by local councils in public parks or as street trees (TL 2009; TIG 2015; GECC 2019; MCC 2019) and were shared species of GU and PR. Another shared species between GU and PR was the widespread shrub or small tree *Pittosporum undulatum*. Australian studies have shown that *P. undulatum* often exploits areas that have had long intervals between fire events and can become invasive in mesic habitats (Gleadow and Narayan 2007). GU and PR sites in the study region are likely to have had long fire intervals due to the mesic environment of GU and the proximity of PR to housing thereby facilitating the success of *P. undulatum* at GU and PR. *Pittosporum undulatum* is also successful due to its tolerance of a wide range of environmental conditions and its exploitation of disturbed areas, which may explain its prevalence in urban vegetation (Mullett and Simmons 1995; Rose and Fairweather 1997; Hammill and Tasker 2010).

The wildland area DS had a significantly higher total species richness than the urban area PR. In particular, I found there to be fewer understorey species present at PR sites than the contrasting wildland assemblages (DS and GU). PR sites were grouped as being parks or other recreational areas that contained significant amounts either native or exotic plants. Given that many PR sites were sports ovals, I would expect understorey richness to be significantly less than that of the surrounding vegetation. Additionally, native overstorey species richness at PR was significantly lower than DS and GU. This was driven by the planting of exotic trees in public parks of the study region. Remnant native overstorey species (typically *Eucalyptus* spp.) have been replaced by exotic trees at PR sites (*Jacaranda mimosifolia, Liquidambar styraciflua, Quercus robur*, *Ulmus minor*, *Ulmus parviflora*).

With the resulting floristic compositional differences between urban and wildland areas at the WUI in the GBMWHA it is important to understand the implications of these differences in a wildfire risk context. In the following chapter, I compare the shoot flammability of key urban and wildland species resulting from the present chapter to better understand wildfire risk at this WUI.

Chapter 3 | Comparative flammability of native and exotic plants at the wildland-urban interface in the Greater Blue Mountains World Heritage Area

3.1 Introduction

A key outcome of the previous chapter was the identification of common plant species at the wildland-urban interface (WUI) in the Greater Blue Mountains World Heritage Area. Importantly, I determined that there were floristic differences in species assemblages between wildland and urban plant communities. In particular, the prevalence of exotic species in household gardens (HG) in urban areas contributed to HG differing floristically from dry sclerophyll woodland (DS) and wet sclerophyll gully (GU) in wildland areas. In addition, species assemblages in public parks and recreational grounds (PR) in urban areas differed floristically from DS in the wildland community. Such broad floristic differences have the potential to lead to differences between wildland and urban plant flammability. In this chapter, I adopt the emerging approach of assessing shoot-level flammability of plant species (Jaureguiberry *et al.* 2011; Calitz *et al.* 2015; Wyse *et al.* 2016; Cui *et al.* 2020a) to identify flammability differences between wildland and urban plants of the WUI.

The flammability of a plant is described as its ability to ignite and sustain a flame when exposed to a fire (Anderson 1970). Measuring the flammability of plant biomass typically involves the ignition of plant material followed by the recording of several attributes whilst the sample burns. Flammability attributes include time to flame (TTF) as a measure of ignitibility, burning time (BT) as a measure of sustainability, maximum combustion temperature (CT) as a measure of combustibility and the percentage of biomass consumed

(BC) in the fire as a measure of consumability (Martin *et al.* 1993; Jaureguiberry *et al.* 2011; Wyse *et al.* 2016; Murray *et al.* 2020). Plant flammability has formerly been assessed at the leaf scale with the hope that leaf scale flammability testing could scale up to predict whole plant flammability. Whilst this approach may still be valid for assessing flammability at the litter bed scale it may not accurately scale up to shoot or whole plant flammability (Fernandes and Cruz 2012; Alam *et al.* 2019). Shoot flammability provides a more accurate method of estimating canopy-level flammability as it incorporates the shoot architecture (branching patterns) of the plant of interest (Jaureguiberry *et al.* 2011). Shoot flammability devices have been used to assess the flammability of vegetation in several ecosystems across the globe (Burger and Bond 2015; Cubino *et al.* 2018; Santacruz-Garcia *et al.* 2019; Cui *et al.* 2020b). However, plant flammability has not yet been assessed at the shoot scale for native Australian plant species. Furthermore, we know little of how the various flammability attributes of plant species relate to each other (Murray et al. 2020).

Exotic plants are typically widespread in urban areas (Smith *et al.* 2006; Kendal *et al.* 2012) and the WUI of this study region is no exception (see Chapter 2). Many highly invasive exotic plants that escaped cultivation were introduced by humans as ornamental plants for domestic gardens (eg. *Lantana camara* (Goncalves *et al.* 2014) and *Ulex europaeus* (Broadfield and McHenry 2019). In some vegetation communities invasive plants increase fuel loads ultimately increasing community-level flammability (Rossiter *et al.* 2003; Berry *et al.* 2011). Yet, in others, exotic invasions reduce overall flammability and facilitate the success of low flammability exotic plants (Pauchard *et al.* 2008). This suggests the effects of exotic plants on fire regimes and flammability dynamics is context-dependent. This study is pioneering for plant flammability research in Australia as it seeks to provide a better understanding of the shoot flammability of native Australian plants as well as common exotic horticultural species from a fire-prone WUI.

In this chapter, I assess shoot-level flammability of 45 locally-abundant plant species at the WUI of the Blue Mountains World Heritage Area. Of the 45 species in the WUI of the region, 26 are locally-abundant in wildland plant communities and 19 are locally-abundant in urban plant communities. The aims of this chapter are:

(i) To determine how common wildland and urban plants differ in ignitability, sustainability, consumability and combustibility;

(ii) To determine how common native and exotic plants of the WUI differ in these four flammability attributes; and

(iii) To examine the role of native and exotic status in driving wildland-urban differences in plant flammability.

3.2 Methods

3.2.1 Study region, species and sampling

The study was performed using species from the WUI of the Greater Blue Mountains World Heritage Area (see Ch. 2 for details of the study region). I selected a total 45 species representing 19 plant families, with 26 wildland species from eight families and 19 urban species from 13 families (Table 3.1). When selecting study species from Chapter 2, I first compiled a list of the most widespread and abundant species. I then elected to prioritise the selection of plants to allow for wide variation in plant traits known to contribute to variation in shoot flammability (Alam *et al.* 2019). A cross-section of species was included from both vegetation types on either side of the wildland-urban interface to provide a representation of native and exotic plants from wildland and urban areas (Table 3.1). Only one exotic plant species, *Lantana camara*, was found at cover abundance values >25% in surveys of wildland sites (see Ch. 2) and I included that species in the qualitative analysis in the present chapter (i.e. in the figures but not in the quantitative analyses as there was no replication of wildland exotic species beyond *L. camara*). Some urban plant species identified in Chapter 2 could not be collected readily in the field. For instance, not enough adult plants could be located, or some plants were only found on inaccessible private property. Thus, I selected replacement species from the same genus or family that occurred commonly in urban areas of the study region (pers. obs.) and which could be purchased from nearby nurseries.

I collected 70 cm sun-exposed shoots from healthy, mature individuals following standardised procedures described in Perez-Harguindeguy *et al.* (2013) and Alam *et al.* (2019). Each 70 cm shoot was collected from a different individual with a minimum of six and a maximum of nine individuals sampled (Perez-Harguindeguy *et al.* 2013). To transport samples back to the laboratory, shoots were placed in large heavy-duty garbage bags and sealed shut to prevent sample moisture loss.

3.2.2 Fuel moisture content

All shoot samples were laid out without overlap on the lab bench to dry down for approximately 18 h to allow for the ignition of a wide range of species (Wyse *et al.* 2016; Cubino *et al.* 2018; Wyse *et al.* 2018; Alam *et al.* 2019). Our initial pilot studies indicated that we did not require the full 24 h drying time as species here tended to be more flammable than NZ species. To record moisture content a 10 cm subsample was taken from each shoot to determine the sample's % moisture content at time of collection (MCC) and at time of burning (MCB). Each subsample's fresh weight (FW) was determined at time of collection

(FWC) and at time of burning (FWB) using a four decimal point benchtop balance (Sartorius Analytical). Subsamples were then dried at 60°C for a minimum of 48 hours to determine subsample dry weight (DW). Using the FW and DW of subsamples, MCC (1) and MCB (2) of shoot samples was calculated using:

$$(1)\ \mathrm{MCC} = \frac{\mathrm{FW\ (at\ collection)} - \mathrm{DW}}{\mathrm{DW}} \times 100\%$$

$$(2)\ \mathrm{MCB} = \frac{\mathrm{FW\ (at\ burn)} - \mathrm{DW}}{\mathrm{DW}} \times 100\%$$

3.2.3 Flammability measurements

To assess shoot-level flammability of WUI plant species we largely followed the methodology described by Jaureguiberry *et al.* (2011) and Wyse *et al.* (2016). This involves the use of a portable, custom-built shoot flammability device (Fig.3.1). Our device was constructed by the UTS machine shop using the specifications of (Jaureguiberry *et al.* 2011) and modified to comply with Australian safety standards. The device has an LPG gas flow that provides heat to radiate through the sample as well as a blowtorch ignition source to ignite the sample. Grill temperatures averaged at approximately 185°C and ranged between 175°C and 190°C. The grill was heated to at least 180°C before flammability testing was commenced and left on for the duration of the experiment. Shoot samples were placed horizontally on the grill for two minutes to simulate the drying effects of a wildfire front (Jaureguiberry *et al.* 2011). During this pre-heating phase of the experiment the sample's length, width and height were recorded. The blowtorch was then switched on for 10 s as per (Jaureguiberry *et al.* 2011). During this 10 s, observers determined at which point the sample ignited to determine time to flaming combustion (TTF). The blowtorch was left on for the full 10 s, even if the sample ignited prior. When the blowtorch was switched off we recorded the duration of flaming combustion as the burning time (BT). Maximum combustion temperature (CT) reached by the sample was measured using a handheld infrared laser thermometer (Digitech QM 7226) and the percentage of biomass consumed (BC) by combustion post ignition was agreed upon by observers visually as per Burger and Bond (2015).

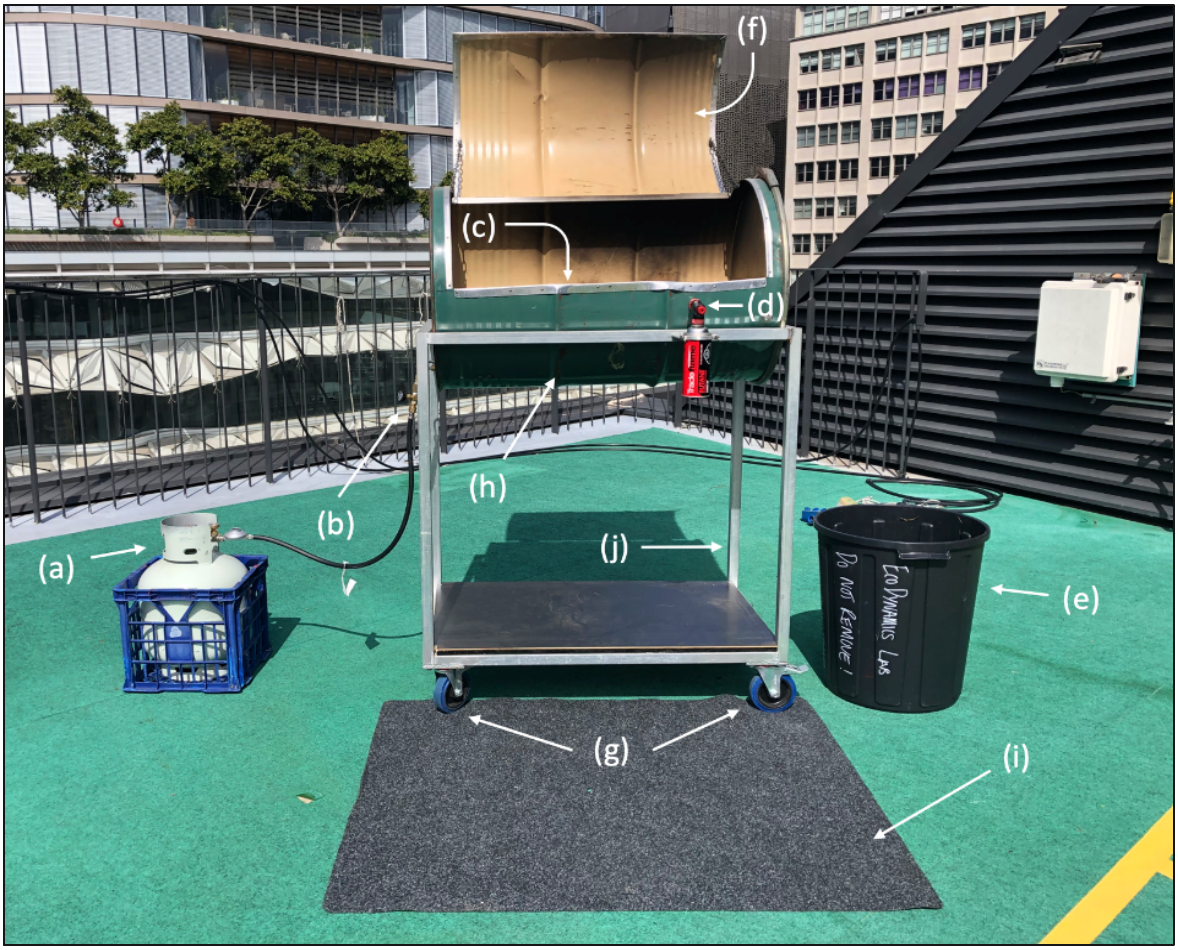

Fig. 3.1 A general diagram of the shoot flammability device used in this research. (a) Gas cylinder, (b) gas flow tap, (c) grill, (d) butane blowtorch, (e) water bath (for extinguishing samples), (f) Wind protection hood, (g) lockable wheels, (h) BBQ barrel, (i) low flammability mat, (j) portable stand.

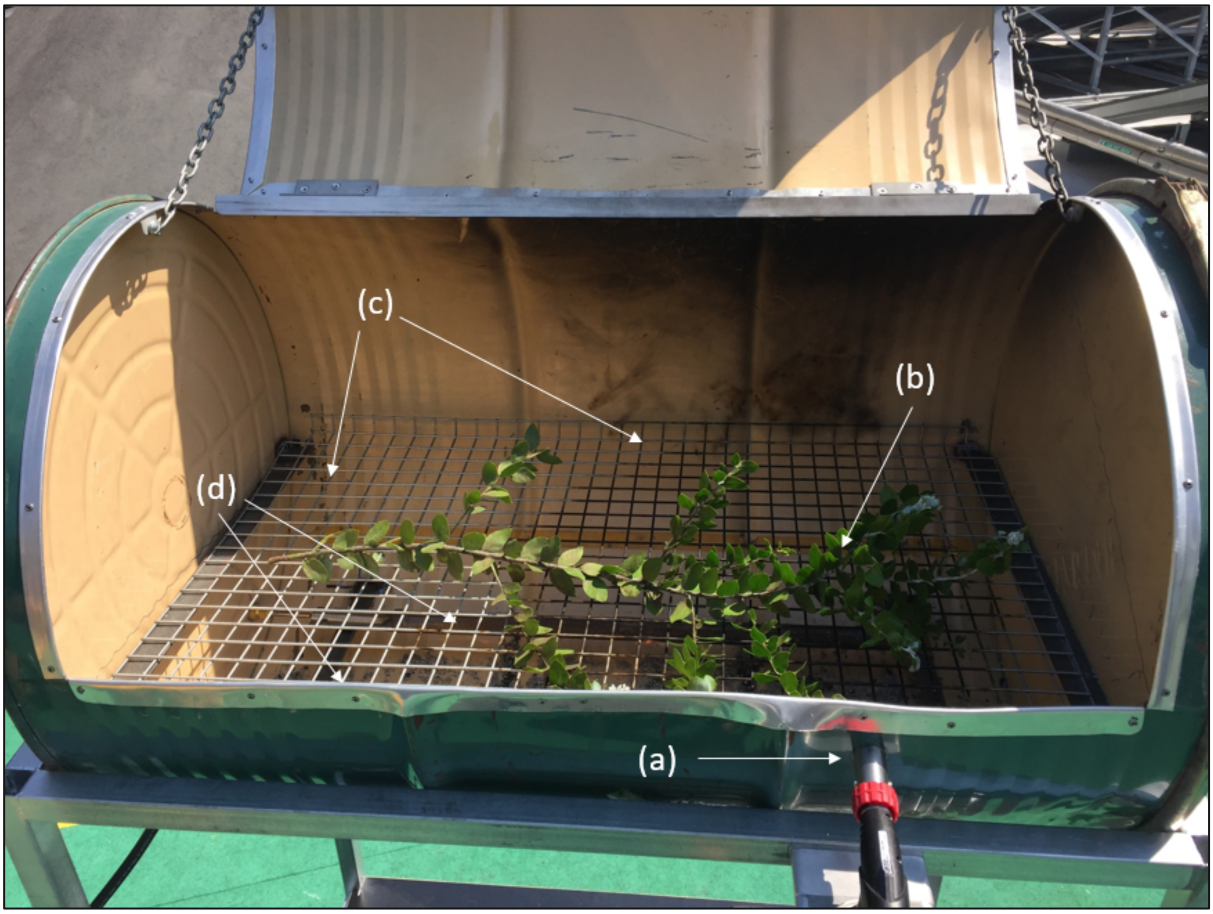

Fig. 3.2 Alternate view of the shoot flammability device. (a) Blowtorch, (b) shoot sample (*Acacia podalyriifolia*), (c) BBQ grill, (d) BBQ burners.

3.2.4 Data analyses

I performed three consecutive sets of analyses that targeted the comparison of each of the four flammability attributes between (1) wildland and urban areas, (2) native and exotic plants, and among (3) wildland native, urban native and urban exotic plant groups. Within each set of analyses, each flammability attribute was treated as a response variable in a separate model with (1) area as a two-level (wildland vs urban) explanatory variable, (2) status as a two-level (native vs exotic) explanatory variable and (3) plant group as a three-level (wildland native vs urban native vs urban exotic) explanatory variable.

The type of analytical model implemented varied depending on the nature of the response variable. General linear models were used for three of the flammability attributes (BT, CT and BC). Log transformation was used for BT, logit transformation for BC, while CT was modelled without transformation. Using observations of TTF (ranging between one and ten seconds) an ignition score was calculated by deducting 11 from all observations and subsequently taking the absolute value to give larger values where samples ignited more quickly. This provided an intuitive understanding of higher ignition scores relating to higher shoot flammability. For samples which did not ignite, a score of zero was assigned. As the ignition score was bounded between zero and ten, quasibinomial modelling was employed after further dividing the ignition score by ten to convert the data to proportions (Crawley 2012).

After model fitting, ANOVA tables were generated for all models using type II (marginal) sums of squares to account for unbalanced number of observations between or among factor levels. For models using plant group as an explanatory variable, significant ANOVA results were followed by post-hoc pairwise tests (with P value adjustment for multiple testing; Benjamini & Hochberg, 1995) to determine which levels of the factor generated the significant finding.

All statistical analyses were performed using R (R core team 2019), using the packages car (Fox and Weisberg 2018), emmeans (Lenth *et al.* 2018) and psych (Revelle 2020).

3.3 Results

A considerable range of interspecific variation was observed in ignition score (Fig. 3.3a), BT (Fig. 3.3b), CT (Fig. 3.3c) and BC (Fig. 3.3d) across all species. In the comparison of wildland-urban plant flammability, I found that BT ($F_{1,43}$ = 4.53, P = 0.04; Fig. 3.4b) and BC ($F_{1,43}$ = 5.26, P = 0.03; Fig. 3.4d) were significantly higher in wildland plants compared with urban plants. However, wildland and urban plants did not differ significantly in terms of either ignition score (χ^2 = 2.62, d.f. = 1, P = 0.10; Fig. 3.4a) or CT ($F_{1,43}$ = 0.0002, P = 0.99; Fig. 3.4c).

The comparison of native-exotic plant flammability demonstrated that exotic plants were significantly less flammable than native plants for three flammability attributes, including BT ($F_{2,41}$ = 11.10, P = 0.0001; Fig. 3.5b), CT ($F_{2,41}$ = 4.69, P = 0.02; Fig. 3.5c) and BC ($F_{2,41}$ = 6.81, P = 0.003; Fig. 3.5d). However, ignition score did not differ significantly between native and exotic plants (χ^2 = 6.13, d.f. = 2, P = 0.05; Fig. 3.5a).

The patterns emerging from the wildland-urban (Fig. 3.4) and native-exotic (Fig. 3.5) comparisons can be further understood by comparing each of the flammability attributes among wildland native, urban native and urban exotic plant groups (Fig. 3.6). In particular, the observed differences in flammability between wildland and urban plants arose because wildland natives were significantly more flammable than urban exotics for BT (t = 4.48, P = 0.0002; Fig. 3.6b) and BC (t = 3.64, P = 0.002; Fig. 3.6d). The lack of significant differences in either BT (t = 0.20, P = 0.80; Fig. 3.6b) or BC (t = 0.43, P = 0.70; Fig. 3.6d) between wildland natives and urban natives underscore the importance of flammability differences between urban exotics and wildland natives in shaping wildland-urban differences in plant flammability. The similarities in wildland and urban plant flammability for ignition score and CT are reflected in the lack of significant differences between wildland native and urban exotic plants for TTF (z = 2.43, P = 0.05; Fig. 3.6a) and CT (t = 1.94, P = 0.09; Fig. 3.6d) and between wildland native and urban native plants for ignition score (z = 0.11, 0.90; Fig. 3.6a) and CT (t = 1.75, P = 0.09; Fig. 3.6d).

Interestingly, within urban areas alone I found that not only were urban native plants significantly more flammable than urban exotic plants for BT (t = 3.81, P = 0.0007; Fig. 3.6b) and BC (t = 2.60, P = 0.02; Fig. 3.6d), mirroring the patterns above for the wildland-urban and native-exotic comparisons, but that urban native plants were significantly more flammable than urban exotic plants for CT (t = 3.06, P = 0.01; Fig. 3.6c). There was no significant difference in ignition score between urban native and urban exotic plants (z = 1.80, P = 0.10; Fig. 3.6a).

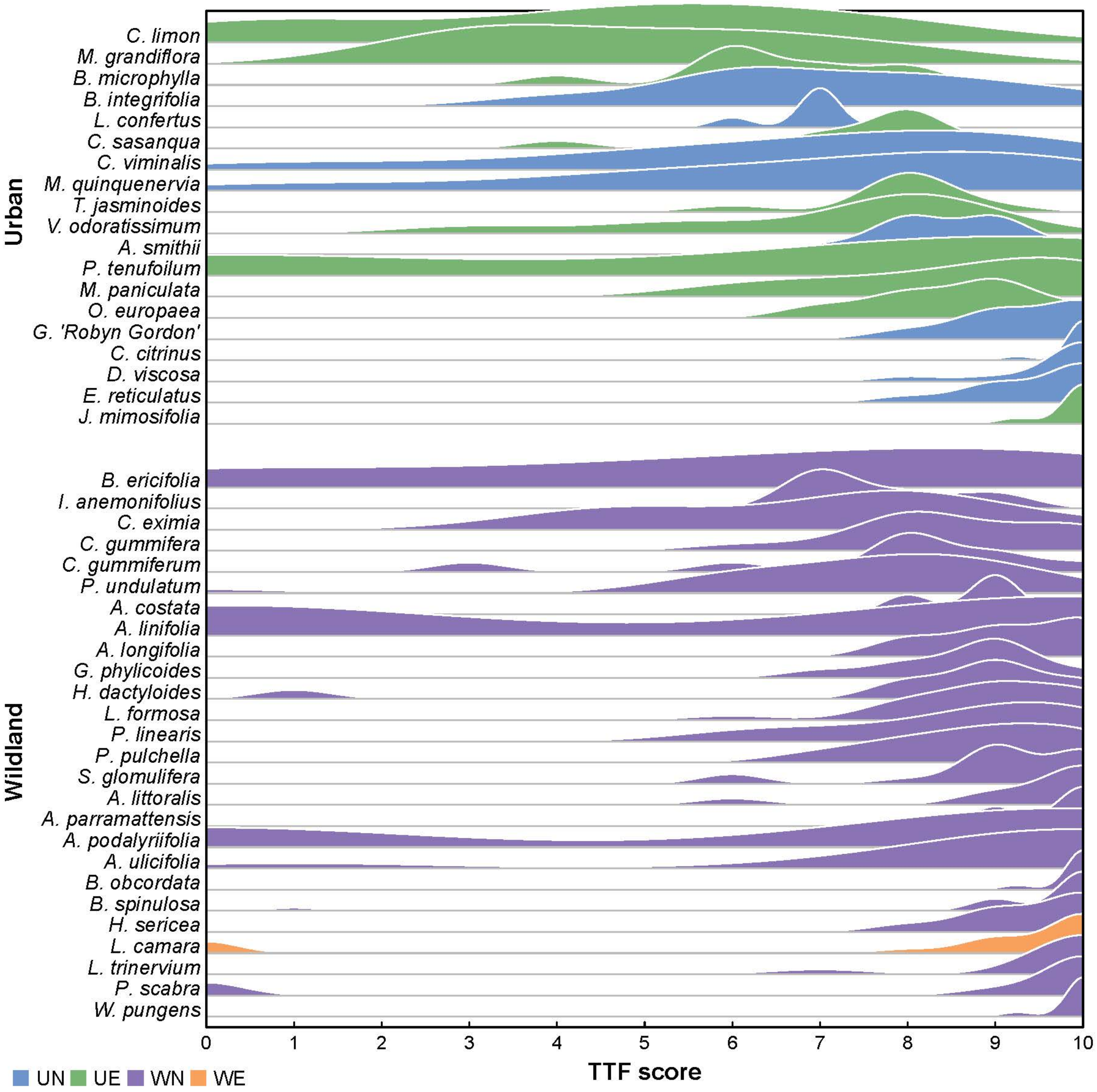

Fig 3.3a Joy plot contains probability densities for time to flame scores (x-axis) across 45 WUI species (y-axis) grouped by WUI status (Urban, Wildland). Within groups species are arranged in ascending order (bottom = most flammable, top = least flammable) in terms of median ignition score. UN = urban native plants; UE = urban exotic plants; WN = wildland native plants; WE = wildland exotic plants.

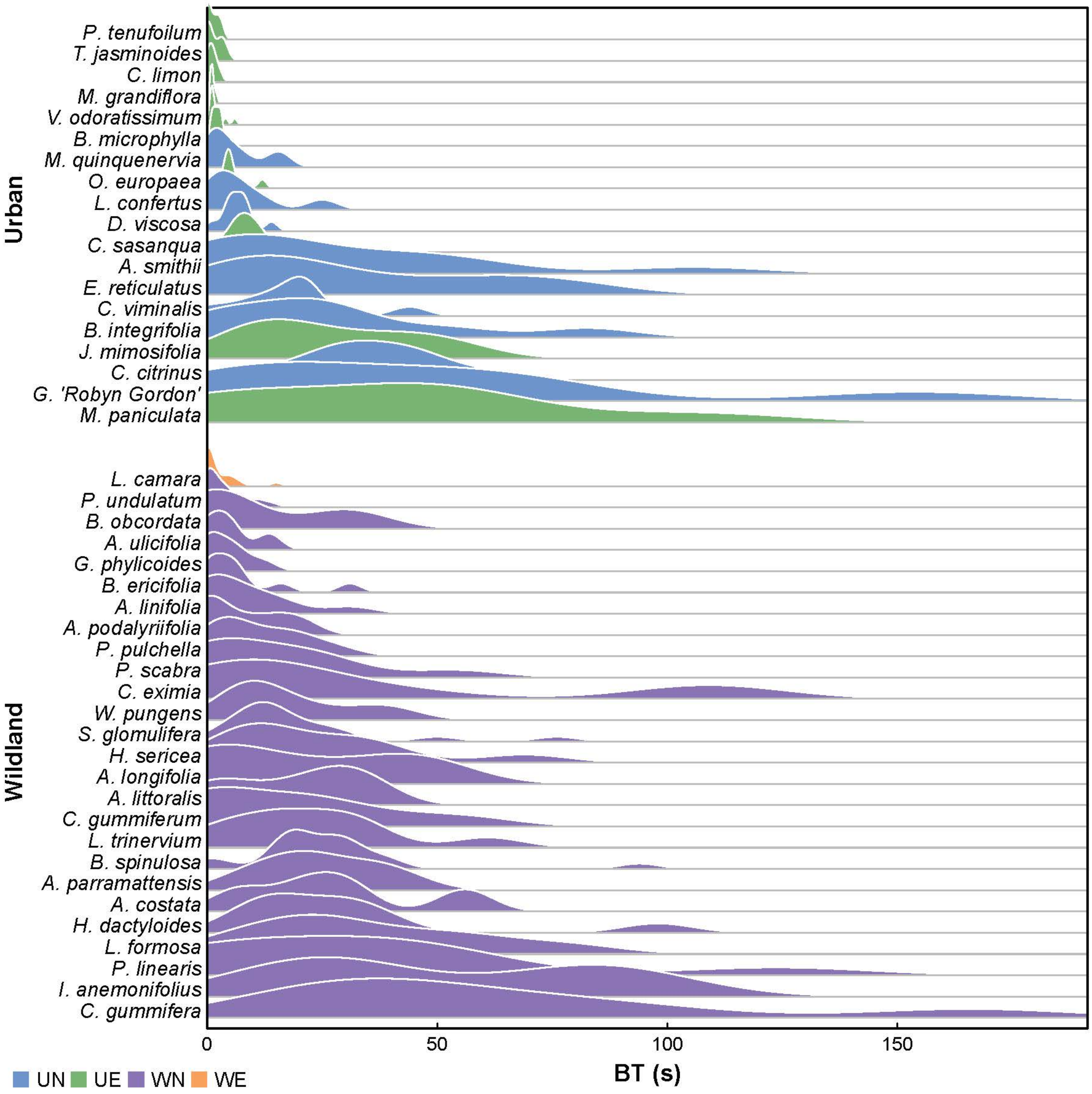

Fig. 3.3b Joy plot for probability densities for values of burn time (x-axis) across WUI species (y-axis) grouped by WUI status (Urban, Wildland). Within groups species are arranged in ascending order (bottom = most flammable, top = least flammable) of median BT values. UN = urban native plants; UE = urban exotic plants; WN = wildland native plants; WE = wildland exotic plants. UN = urban native plants; UE = urban exotic plants; WN = wildland native plants; WE = wildland exotic plants.

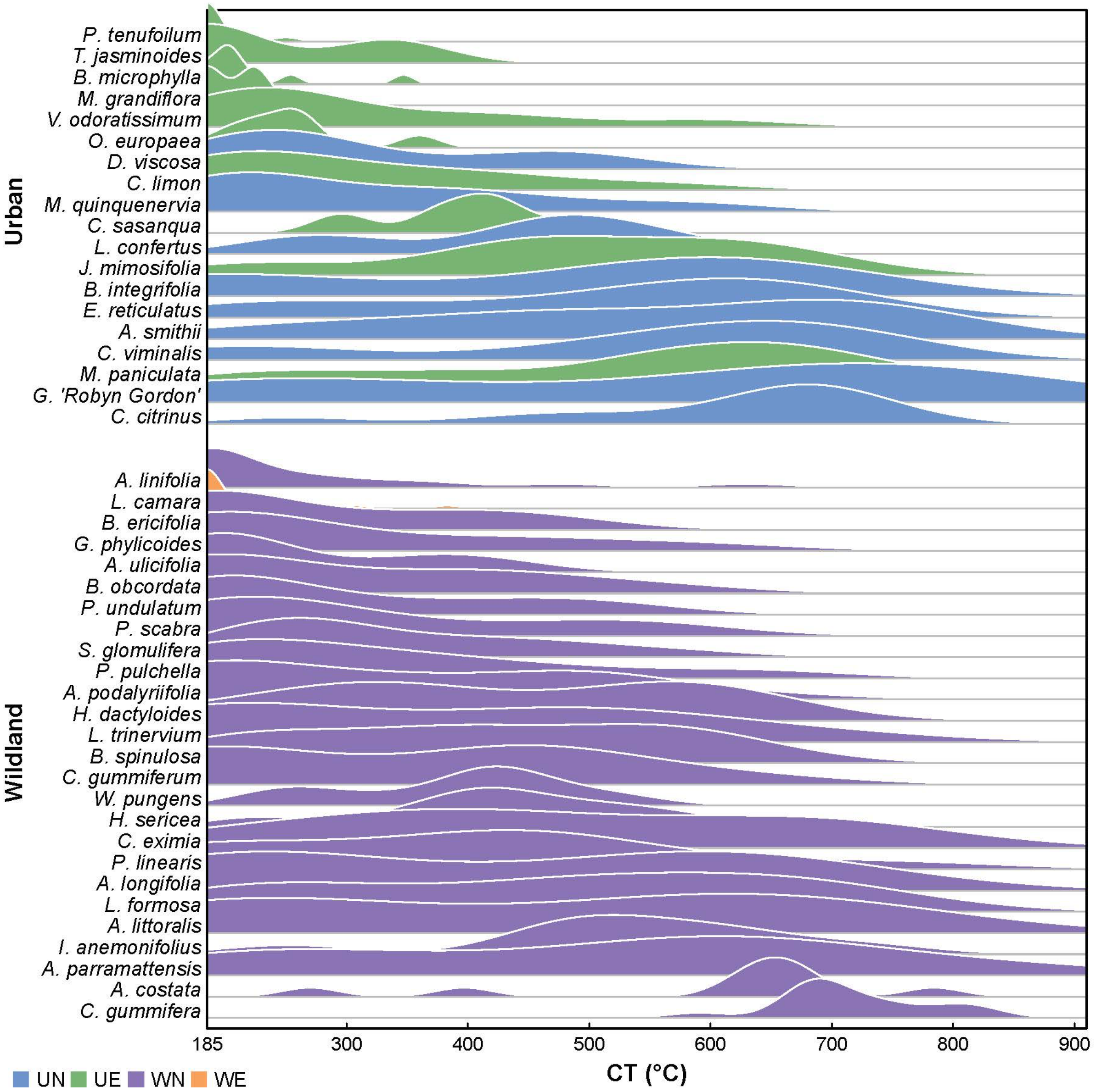

Fig. 3.3c Joy plot contains probability densities for combustion temperatures (x-axis) across WUI species (y-axis) grouped by WUI status (Urban, Wildland). Within groups species are in ascending order (bottom = most flammable, top = least flammable) of median CT values. UN = urban native plants; UE = urban exotic plants; WN = wildland native plants; WE = wildland exotic plants.

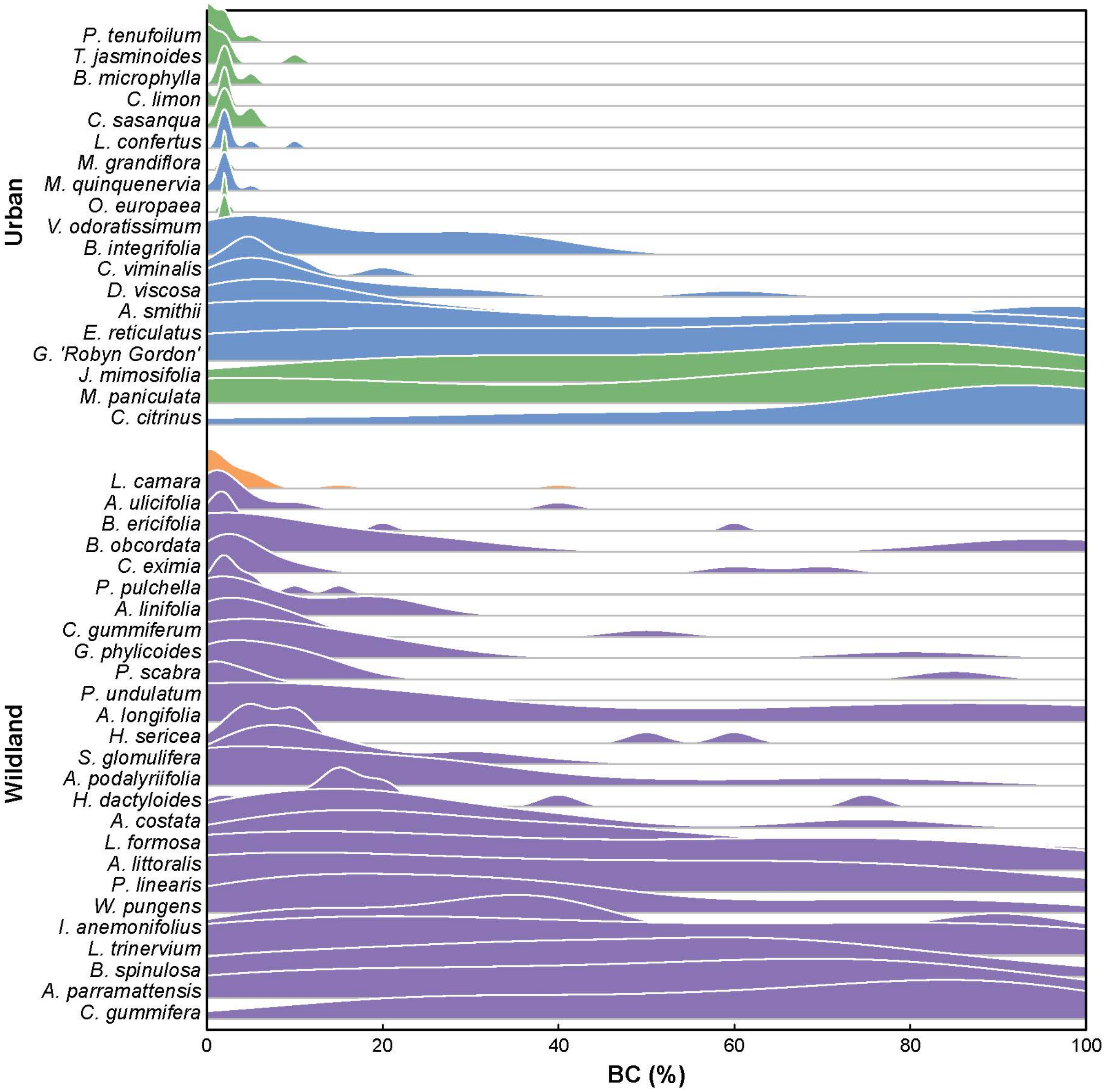

Fig. 3.3d Joy plots contain probability densities for biomass consumed (x-axis) across 45 WUI species (y-axis), grouped by WUI status (Urban, Wildland). Within groups species are arranged in ascending order (bottom = most flammable, top = least flammable) of median BC values. Green = urban-exotic species; blue = urban-native species; orange = wildland-exotic species; purple = wildland-native species.

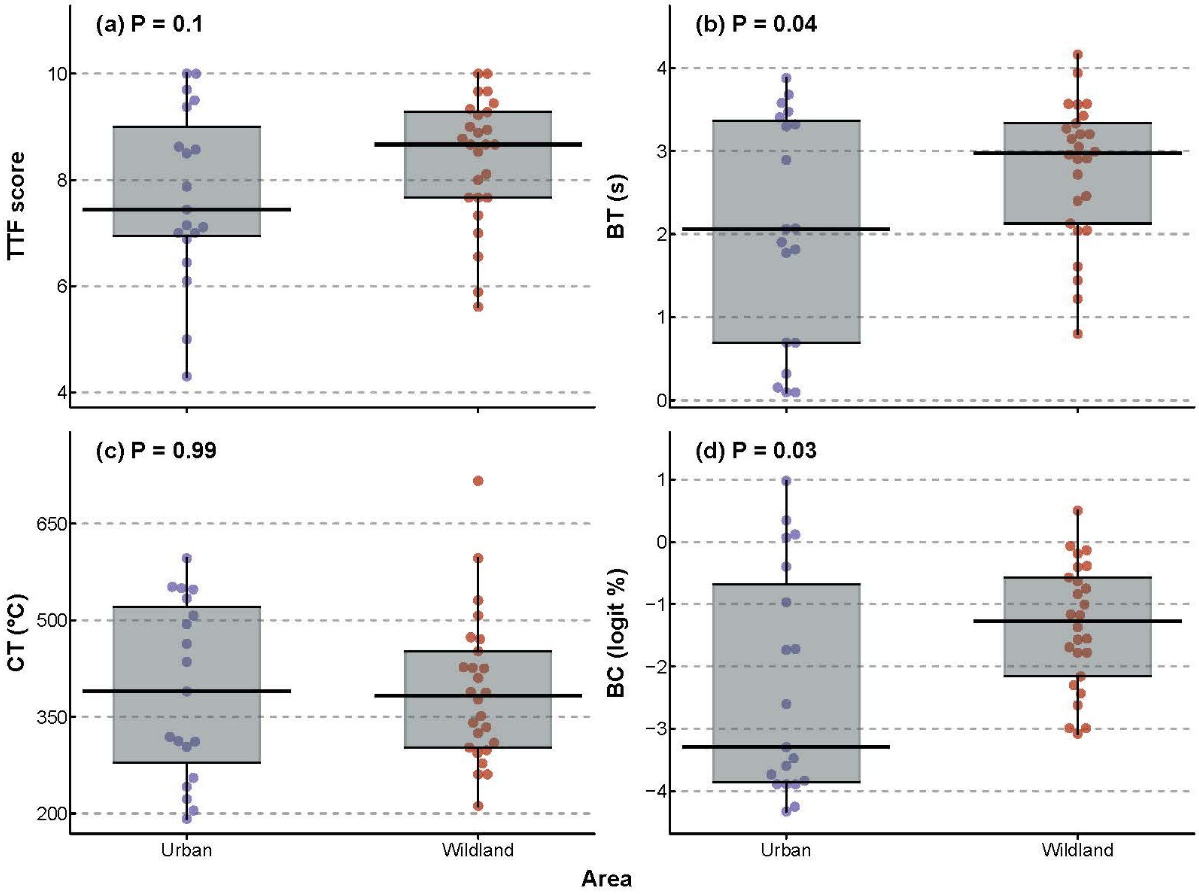

Fig. 3.4 Boxplots for **(a)** mean time to flame (ignition) score, **(b)** mean burn time (BT), **(c)** mean combustion temperature (CT), **(d)** mean biomass consumed (BC) across all species. Box = interquartile range; line across box = mean; whiskers = range of values; points on the figures = species values; urban species = blue, wildland species = red; P = statistical significance.

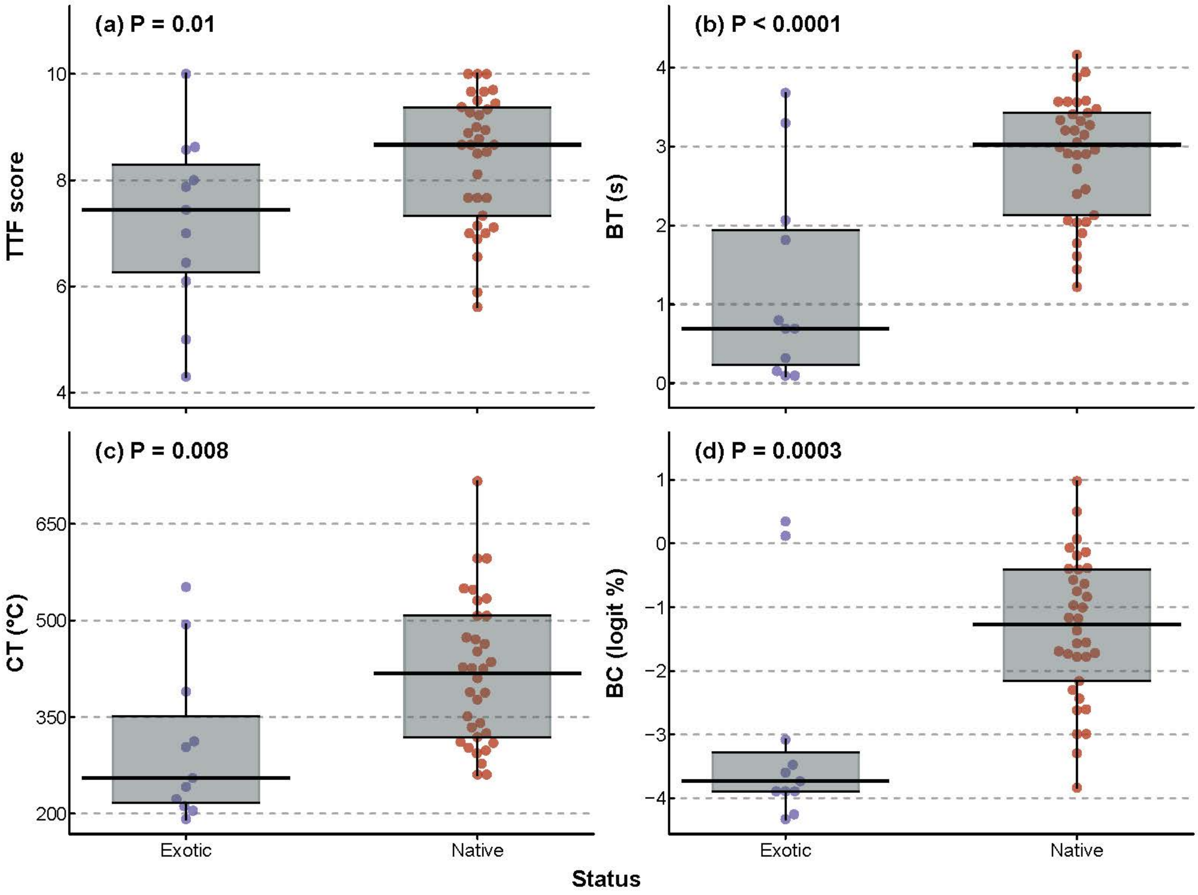

Fig. 3.5 Boxplots for **(a)** mean time to flame (ignition) score, **(b)** mean burn time (BT), **(c)** mean combustion temperature (CT), **(d)** mean biomass consumed (BC) across all species. Box = interquartile range; line across box = mean; whiskers = range of values; Points on the figures = species values; exotic species = blue, native species = red; P = statistical significance.

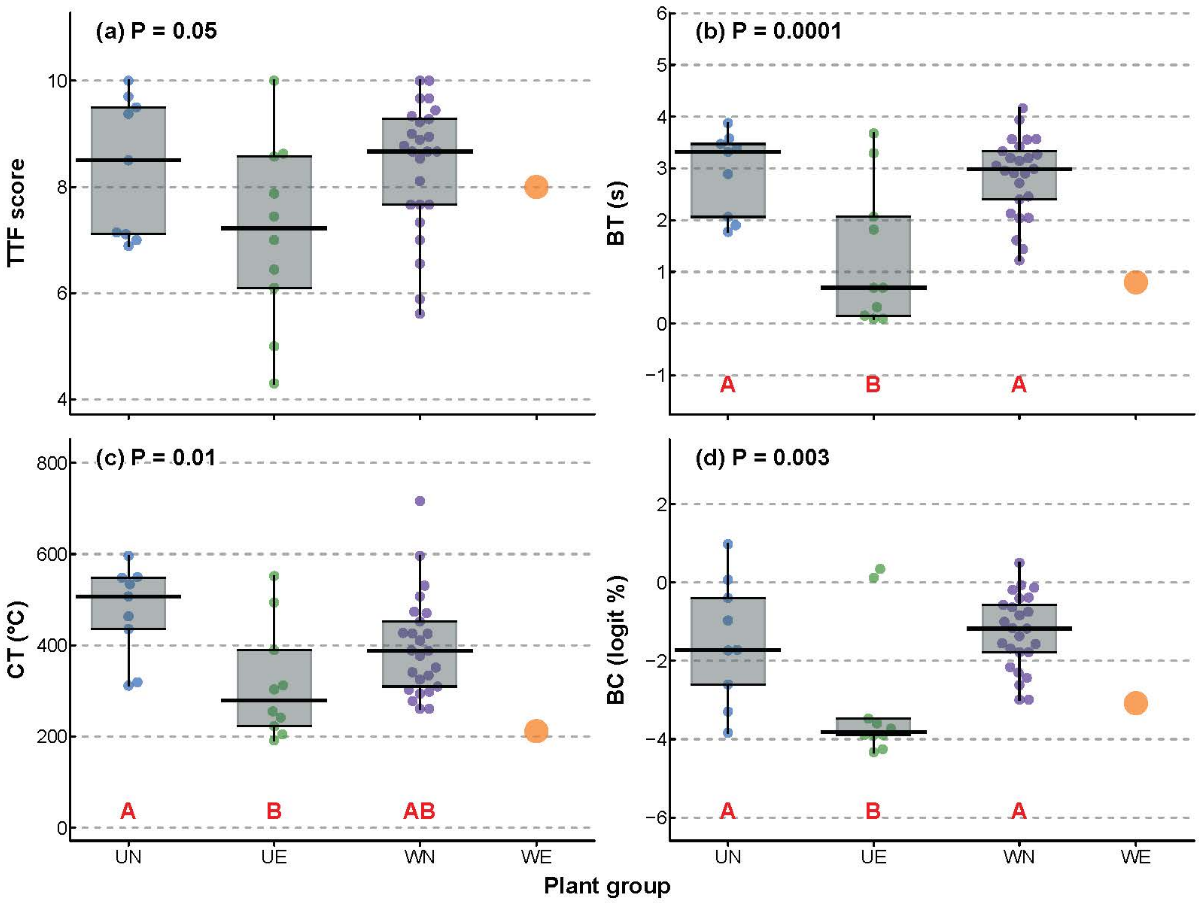

Fig. 3.6 Boxplots for **(a)** mean time to flame (ignition) score, **(b)** mean burn time (BT), **(c)** mean combustion temperature (CT), **(d)** mean biomass consumed (BC) across all species. Box = interquartile range; line across box = mean; whiskers = range of values; Points on the figures = species values; Plant groups are as follows, UN = urban native species (blue); UE = urban exotic species (green); WN = wildland native species (purple); WE = wildland exotic species (orange); P = statistical significance. Note: WE category has one value, due to their only being one species (*L. camara*) in this category.

3.4 Discussion

Given the paucity of literature examining plant flammability at WUIs, in this chapter, I investigated differences in flammability between common plants from urban and wildland areas of this fire-prone WUI. I found significant differences in flammability between common urban and wildland plants. Specifically, common plants from the wildland side of the WUI were more flammable than urban plants, with longer burning times and more biomass consumed during shoot flammability testing. For example, the most flammable species was the wildland eucalypt, *Corymbia gummifera*, which ignited relatively quickly, burnt long, hot and consumed a high proportion of biomass. Interestingly, in terms of BT, four Proteaceous species were the next four most flammable species (i.e. *Isopogon anemonifolius*, *Persoonia linearis*, *Lambertia formosa*, *Hakea dactyloides*). This difference in shoot flammability between urban and wildland plants was underpinned by exotic plants on the urban side of this WUI. Specifically, urban exotic plants, *Pittosporum tenufolium*, *Trachelospermum jasminoides*, and *Citrus limon* were the least flammable in terms of BT and BC.

Urban exotic plants were less flammable than native wildland plants, with significantly shorter burning times and significantly less biomass consumed. Exotic plants can have mixed effects on flammability dynamics, with some exotic invaders being more flammable and others less flammable (Rossiter *et al.* 2003; Brooks *et al.* 2004; Cubino *et al.* 2018). In work from Wyse *et al.* (2016), and contrary to my findings, many exotic species were more flammable than indigenous NZ species due to these highly flammable exotic species originating from fire-prone vegetation communities. Among these species were two Australian native species, and one wildland native species (*Hakea sericea*) from this WUI (Wyse *et al.* 2016). This suggests that the native-exotic plant flammability dynamic is context-dependent. There's evidence to suggest that fire-prone plant communities may contain pyrophilic native plants that may have inherited traits to enhance fire frequency or intensity in an effort to kill other less pyrophilic plants (Bond and Midgley 1995). There is also evidence to suggest that individuals from areas with recurrent fire are significantly more flammable than individuals of the same species from fire-free areas (Pausas *et al.* 2012). The theory that plants in fire-dependent plant communities have evolved traits that have enhanced their flammability was first introduced by Mutch (1970) and has been under some debate in flammability literature (Bond and Midgley 1995; Schwilk and Kerr 2002; Bowman *et al.* 2014; Pausas *et al.* 2017). Whilst this thesis did not set out to address the hypothesis of Mutch (1970) there is some evidence to suggest that native wildland plants from wildland vegetation in this fire-prone WUI are highly flammable and support the idea of recurrent fire selecting for higher plant flammability (Pausas *et al.* 2017). Considering that this WUI sits within fire-prone eucalypt dominant vegetation communities (Hammill and Tasker 2010) and many of the exotic urban plants in this WUI are indigenous to ecosystems that have little to no fire (e.g. *Camellia sasanqua*), it is perhaps expected that exotic urban plants would be less flammable than native wildland plants in the Blue Mountains.

Interestingly, urban native plants were more flammable than urban exotic plants in terms of sustainability, consumability and combustibility but no significant differences were found in terms of ignitability. These first three components relate to fire-residence time and fire intensity thus, these findings have serious implications for urban plant selection at Australian WUIs. It is common in flammability 'grey literature' (brochures, pamphlets, blog posts and other web pages) to find resources that contain plant lists with many Australian native plants listed as low flammability alternatives for urban garden design (Murray *et al.* 2020). However, in most cases these 'grey literature' resources contain qualitative measures of flammability and lack formal flammability testing. As hypothesised in Murray *et al.* (2020) and brought to light in some of our unpublished work we have shown that some of these resources contradict each other. This leads to species being recommended in low flammability plant lists when they are in fact high flammability species (e.g. *Dodonaea viscosa*). In line with my results, at the leaf scale, Ganteaume et al. (2013) showed that some WUI hedge species in urban areas were highly flammable. Given the presence of high flammability natives in urban and wildland areas of the WUI of the study region, this study highlights the paucity of research in plant flammability at the WUI and in Australian ecosystems as a whole. This calls for formal flammability testing of horticultural native species before claims of low flammability can be made. Additionally, at larger scales, a modelling approach in line with Zylstra *et al.* (2016) that takes into account floristic composition and flammability attributes of plants (Jaureguiberry *et al.* 2011) will improve wildfire modelling and thus wildfire hazard management in fire-prone areas.

Considering that urban native plants in this WUI were more flammable than urban exotic plants, does not detract from the value of using urban native plants in urban areas to promote biodiversity. Invasion of vegetation communities by exotic plants is a major threat to biodiversity globally (Vavra *et al.* 2007). Urban areas are typically high in exotic plant species richness and abundance (Smith *et al.* 2006; Acar *et al.* 2007) as is true for the study region (See Ch. 2). Exotic ornamental plants have the potential to escape cultivation and become invasive in surrounding native plant communities (Bell *et al.* 2003). For example, the ornamental hedge species from tropical America, *Lantana camara* has invaded many areas of Africa, Asia, and Oceania (including Australia) and has deleterious impacts on the recruitment and survival of native plant species (Goncalves *et al.* 2014). Similarly, introduced as an ornamental hedge plant in Australia, *Ulex europaeus* (Gorse) is a weed of national significance as it forms dense thickets in bushland and agricultural land in Australia and across the globe (Broadfield and McHenry 2019). Gorse has also been reported as a high flammability plant due to its retention of dead material (Dent *et al.* 2019) and has the potential to increase the flammability of early successional vegetation communities (Perry *et al.* 2014). Conversely, native plants in urban areas can provide important habitat and food resources for native animals. Species richness and abundance of bees was positively correlated native plant dominated (> 48% cover) in urban domestic gardens in the United States (Pardee and Philpott 2014). In Australia, large hollow-bearing native trees are a keystone resource in urban parks as they provide critical habitat for birds and subsequently increase bird diversity in urban areas (Stagoll *et al.* 2012). The native plants in this study that were the lowest in flammability were the Fabaceous shrubs, *Acacia ulicifolia* and *Bossiaea obcordata* and showed BT and BC values synonymous with low flammability urban exotic plants (i.e. *Buxus microphylla* and *Viburnum odoratissimum*). Whilst there is still much work to be done on identifying low flammability native plants, these two species may provide a starting point.

In this chapter, I demonstrated that the large range of interspecific shoot-level flammability reported in other biomes across the globe (Calitz *et al.* 2015; Wyse *et al.* 2016; Cubino *et al.* 2018; Santacruz-Garcia *et al.* 2019; Cui *et al.* 2020a) is also true for native and exotic plants in this Australian WUI system. This study is novel for Australian vegetation as it takes a recently standardised approach to measuring plant flammability (Jaureguiberry *et al.* 2011; Burger and Bond 2015; Wyse *et al.* 2018; Santacruz-Garcia *et al.* 2019; Cui *et al.* 2020a) with strong correlations of observed plant flammability in the field (Wyse *et al.* 2016) to quantify plant flammability in fire-prone WUI. In the following chapter, I provide an analysis of shoot morphological traits with measured shoot flammability attributes to better understand the mechanisms that drive variation among species in each of the flammability attributes.

Table 3.1 Species, family, status (native = N, exotic = E), WUI (wildland = W, urban = U), and WUI status (WN, WE, UN, UE) for the 45 study species. * = species that were widespread in WUIs but were replacements of other study species.

Species	**Species name**	**Family**	**WUI**	**Status**	**WUI status**	**Vegetation**
Acacia linifolia	*A. linifolia*	Fabaceae	W	N	WN	GU
Acacia longifolia	*A. longifolia*	Fabaceae	W	N	WN	GU
Acacia parramattensis	*A. parramattensis*	Fabaceae	W	N	WN	DS
Acacia podalyriifolia	*A. podalyriifolia*	Fabaceae	W	N	WN	DS
Acacia ulicifolia	*A. ulicifolia*	Fabaceae	W	N	WN	DS
Acmena smithii	*A. smithii*	Myrtaceae	U	N	UN	HG*
Allocasuarina littoralis	*A. littoralis*	Casuarinaceae	W	N	WN	GU
Angophora costata	*A. costata*	Myrtaceae	W	N	WN	GU
Banksia ericifolia	*B. ericifolia*	Proteaceae	W	N	WN	GU
Banksia integrifolia	*B. integrifolia*	Proteaceae	U	N	UN	HG
Banksia spinulosa	*B. spinulosa*	Proteaceae	W	N	WN	DS
Bossiaea obcordata	*B. obcordata*	Fabaceae	W	N	WN	DS
Buxus microphylla	*B. microphylla*	Buxaceae	U	E	UE	PR
Callistemon citrinus	*C. citrinus*	Myrtaceae	U	N	UN	HG
Callistemon viminalis	*C. viminalis*	Myrtaceae	U	N	UN	HG
Camellia sasanqua	*C. sasanqua*	Theaceae	U	E	UE	HG
Ceratopetalum gummiferum	*C. gummiferum*	Cunoniaceae	W	N	WN	GU*

Species	**Species name**	**Family**	**WUI**	**Status**	**WUI status**	**Vegetation**
Citrus limon	*C. limon*	Rutaceae	U	E	UE	HG
Corymbia eximia	*C. eximia*	Myrtaceae	W	N	WN	DS
Corymbia gummifera	*C. gummifera*	Myrtaceae	W	N	WN	DS
Dodonaea viscosa	*D. viscosa*	Sapindaceae	U	N	UN	PR*
Elaeocarpus reticulatus	*E. reticulatus*	Elaeocarpaceae	U	N	UN	PR*
Grevillea banksii x bipinnatifida 'Robyn Gordon'	*G. 'Robyn Gordon'*	Proteaceae	U	N	UN	HG
Grevillea phylicoides	*G. phylicoides*	Proteaceae	W	N	WN	GU
Hakea dactyloides	*H. dactyloides*	Proteaceae	W	N	WN	GU
Hakea sericea	*H. sericea*	Proteaceae	W	N	WN	DS
Isopogon anemonifolius	*I. anemonifolius*	Proteaceae	W	N	WN	DS*
Jacaranda mimosifolia	*J. mimosifolia*	Bignoniaceae	U	E	UE	HG
Lambertia formosa	*L. formosa*	Proteaceae	W	N	WN	DS
Lantana camara	*L. camara*	Verbenaceae	W	E	WE	GU
Leptospermum trinervium	*L. trinervium*	Myrtaceae	W	N	WN	GU
Lophostemon confertus	*L. confertus*	Myrtaceae	U	N	UN	PR
Magnolia grandiflora	*M. grandiflora*	Magnoliaceae	U	E	UE	HG
Melaleuca quinquenervia	*M. quinquenervia*	Myrtaceae	U	N	UN	PR
Murraya paniculata	*M. paniculata*	Rutaceae	U	E	UE	HG
Olea europaea	*O. europaea*	Oleaceae	U	E	UE	HG
Persoonia linearis	*P. linearis*	Proteaceae	W	N	WN	DS*

Species	**Species name**	**Family**	**WUI**	**Status**	**WUI status**	**Vegetation**
Petrophile pulchella	*P. pulchella*	Proteaceae	W	N	WN	DS
Pittosporum tenuifolium	*P. tenufoilum*	Pittosporaceae	U	E	UE	HG
Pittosporum undulatum	*P. undulatum*	Pittosporaceae	W	N	WN	GU
Pultenaea scabra	*P. scabra*	Fabaceae	W	N	WN	DS
Syncarpia glomulifera	*S. glomulifera*	Myrtaceae	W	N	WN	DS
Trachelospermum jasminoides	*T. jasminoides*	Apocynaceae	U	E	UE	HG
Viburnum odoratissimum	*V. odoratissimum*	Adoxaceae	U	E	UE	HG
Woollsia pungens	*W. pungens*	Ericaceae	W	N	WN	DS

Chapter 4 | Shoot traits do not adequately explain differences in shoot level flammability between natives and exotics of dominant WUI plants

4.1 Introduction

In Chapter 3, I assessed the relative flammability of common and widespread plant species at the WUI in the Blue Mountains. In this chapter, I seek to explore relationships between shoot traits and shoot flammability of these WUI species. The overall aim is to understand how variation in plant functional traits underpin observed differences in shoot flammability between native and exotic species from urban and wildland sides of this fire-prone WUI.

Plant functional traits are the morphological, physiological and phenological features of a plant that determine its response to environmental changes (Perez-Harguindeguy *et al.* 2013). Plant functional traits are well researched worldwide (Kattge *et al.* 2011), as ecologists attempt to understand ecological and evolutionary patterns and processes that determine the distributions of plant species in past, present and future environmental conditions (Perez-Harguindeguy *et al.* 2013). Within fire ecology, trait-based experiments seek to use easy-to-measure traits that correlate well with wildfire metrics to enable a better understanding of wildfire hazards, evolutionary life-histories and exotic species invasions in a changing world. Global patterns in plant functional trait variation have been shown through the leaf economics spectrum (Wright *et al.* 2004) indicating that functional traits could provide a means to understand and predict changes to ecological communities in a rapidly changing world.

As a plant functional trait, flammability is broadly defined as the likelihood of a plant to ignite and sustain combustion when exposed to an ignition source (Anderson 1970). Shoot-level flammability is recognised as a core functional plant trait (Perez-Harguindeguy *et al.* 2013) and has the potential to scale up to estimating the relative flammability of whole plants (Wyse *et al.* 2016). With increasing research in shoot-level flammability some trait-based patterns have emerged. In particular, Burger and Bond (2015) showed that the most important variables for determining the proportion of biomass burnt (consumability) in Cape shrubland species was the proportion of fine fuel (fuel < 3 mm) and the percentage of dead material retained by plants. A simple measurement of total biomass appears to correlate well with flammability in grassland ecosystems (Simpson *et al.* 2016; Cubino *et al.* 2018). Alam *et al.* (2019) showed that whilst leaf flammability was decoupled from shoot flammability there were several leaf traits that may be drivers of shoot flammability. Particularly, lignin content, leaf dry matter content (LDMC) and leaf thickness correlated well with overall flammability across 60 common and widespread NZ indigenous and exotic species (Alam *et al.* 2019). Leaf lignin content and LDMC correlated positively with overall flammability and leaf thickness had a negative relationship with the ignitability component (Alam *et al.* 2019). Calitz *et al.* (2015) struggled to find robust relationships with plant functional traits and shoot flammability but did find small-leaved plants with high “twigginess” (assessed visually) to have high relative flammability across the Eastern Cape biomes of South Africa. Santacruz-Garcia *et al.* (2019) assessed plant functional traits as a proxy for measuring shoot flammability in the Argentine Chaco Region. They found that functional traits alone were relatively good at predicting the flammability categories that species may fall into (low, moderate, high) but failed to predict the most flammable species of the 11 woody species (Santacruz-Garcia *et al.* 2019). Given that many of Australia’s ecosystems are considered among the most fire-prone communities in the world (Bowman 2003; Gill and Zylstra 2005) and the recent catastrophic fire events (Nolan *et al.* 2020) it is integral for future management strategies that we appropriately understand the flammability of plants at WUIs and the traits that may explain mechanisms (Murray et al. 2020).

The advantage of assessing plant flammability at the shoot level is the preservation of branching architecture of shoot samples to be ignited (Jaureguiberry *et al.* 2011). Plants that are highly branched are thought to have evolved this branching pattern under high grazing pressure from vertebrates (Archibald and Bond 2003; Perez-Harguindeguy *et al.* 2013) (Archibald and Bond 2003; Perez-Harguindeguy *et al.* 2013). Fire ecologists have also been attracted to branching architecture as a possible trait evolved in high flammability plants in fire prone plant communities (Schwilk 2003). The theory stems from the contentious hypothesis of Mutch (1970), which states that species persisting in fire-prone ecosystems may have evolved traits that enhance the flammability of their community. Yet, despite this contention few studies explore canopy architecture and flammability in woody plant species (but see Schwilk 2003). In this chapter, I assess the importance of canopy architecture and other shoot traits on the comparative flammability of native and exotic plant species at the WUI. Canopy architecture may be defined as the degree of ramification (DR) of a branch (Perez-Harguindeguy *et al.* 2013), and in addition to DR I assess the influence of and bulk density, shoot water content and shoot dry biomass. Plants that are highly branched are thought to have evolved this branching pattern under high grazing pressure from vertebrates (Archibald and Bond 2003; Perez-Harguindeguy *et al.* 2013). Fire ecologists have also been attracted to plant architecture as a possible trait evolved in high flammability plants in fire-prone plant communities (Schwilk 2003). The theory stems from the contentious hypothesis of Mutch (1970), which states that species persisting in fire-prone ecosystems may have evolved traits that enhance the flammability of their community. Yet, despite this contention there have been few studies that have explore canopy architecture and flammability in woody species (but see Schwilk 2003). In this chapter, I assess the importance of canopy architecture and other shoot traits on the comparative flammability of native and exotic plant species at the WUI. I ask, if shoot traits can explain the interspecific variation in shoot flammability observed in Chapter 3 and what contributions shoot traits have on observed differences in flammability between the three plant groups (wildland-native, urban-native and urban-exotic)?

4.2 Methods

4.2.1 Shoot flammability

The Jaureguiberry *et al.* (2011) method burns 70 cm shoots that retain a plant's natural branching pattern or ramification. The procedure for igniting shoot samples is based on protocols followed by (Wyse *et al.* 2016), using the device developed by Jaureguiberry *et al.* (2011) modified to meet Australian safety standards. All flammability data used in this chapter was collected following the methodology outlined in Chapter 3. The flammability attributes examined here include shoot ignitability (ignition score), sustainability (BT), combustibility (CT) and consumability (BC).

4.2.2 Degree of ramification

To measure shoot architecture I used degree of ramification (DR), a core functional plant trait (Perez-Harguindeguy *et al.* 2013). This is the number of branches (or nodes) along the main stem line per metre of branch. I measured the length of branches to nearest mm and counted the number of branches per main to stem to calculate DR (number of branches per m). I measured this trait on a total of nine individuals per species from three individuals across three sites. There was a minimum of 5 replicates per individual per species as recommended by Perez-Harguindeguy *et al.* (2013). Some measurements were taken on live plants in the field and others were taken on shoots prior to burning.

4.2.3 Bulk density

Bulk density is a metric that takes into account the dry biomass (kg) and the gross volume of the sample, assuming the shape of the sample is conical with an elliptical base (equation 1). Bulk density of shoot-level flammability was described by Wyse *et al.* (2018) and successfully replicated in other shoot flammability studies (Cubino *et al.* 2018; Wyse *et al.* 2018). The dry biomass per sample was calculated from the fresh weight (FWB) of the sample measured prior to burning (Wyse *et al.* 2018). The subsample's MC is used as a calibration factor that converts FWB to an approximate dry biomass (Wyse *et al.* 2018). The length, width, and height of sample were taken whilst sample was pre-heating during the burning phase of the experiment (See Ch. 3).

$$BD = 12 \times \frac{dry\ biomass}{\pi \times width \times height \times length} \quad (1)$$

4.2.4 Shoot water content

Typically, the hydration of a sample is expressed as fuel moisture content (FMC). This is the amount of water within a shoot sample as a proportion of the sample's dry mass (Perez-Harguindeguy *et al.* 2013). Note that SWC (used here) is hydration as a proportion of a sample's FW (2).

$$SWC = \frac{FW - DW}{FW} \times 100 \ (2)$$

4.2.5 Data analyses

Principal components analysis (PCA) was implemented to transform the shoot trait data (centred and scaled) into a smaller number of underlying constructs (Tabachnik and Fidell 1989). The component loading matrix was rotated (varimax rotation) following extraction to achieve simple solution. This method generated two major independent principal components (PC1, PC2), that together explained 68.2% of variation in the shoot trait variables (Fig. 1). The first PC (PC1) accounted for nearly 34.5% of the variance in the shoot trait data set, and had a high positive loading relative to the other traits on shoot dry weight (Table 1). In contrast, PC2, which accounted for 33.7% of variation in shoot traits, primarily described shoot water content (Table 4.1). There was no significant loading for plant morphological traits DR and BD on either of the PC axes.

I used multiple regression with model simplification to quantify the relationships between each of the flammability attributes (response variables in separate models) and PC1, PC2 and status (explanatory variables). In multiple regressions, I was particularly interested in the unique contribution of each explanatory variable (e.g. Murray and Hose 2005). This contribution reflects the variation in the response variable that is accounted for by each explanatory variable over and above the effects of the other explanatory variables in the model. To achieve this, we built minimum adequate models (MAMs) for each flammability attribute that also included the two-way interactions of PC1 x status and PC2 x status. The building of MAMs proceeded by fitting the maximal model first (i.e. all explanatory variables), followed by testing the significance of each variable when removed from the maximal model. The variable with the lowest non-significant change in deviance was then dropped and a reduced model was then fitted with all remaining variables. This procedure was continued until a subset of explanatory variables was obtained, each producing a significant change in deviance when removed from a reduced model (Crawley 2012).

The type of analytical model implemented varied depending on the nature of the response variable. General linear models were used for three of the flammability attributes (BT, CT and BC). Log transformation was used for BT, logit transformation for BC, while CT was modelled without transformation. Using observations of TTF (ranging between one and ten seconds) an ignition score was calculated by deducting 11 from all observations and subsequently taking the absolute value to give larger values where samples ignited more quickly. This provided an intuitive understanding of higher ignition scores relating to higher shoot flammability. For samples which did not ignite, a score of zero was assigned. As the ignition score was bounded between zero and ten, quasibinomial modelling was employed after further dividing the ignition score by ten to convert the data to proportions (Crawley 2012).

After model fitting, where status was retained as a significant term, ANOVA tables were generated for all models using type II (marginal) sums of squares to account for unbalanced number of observations between or among factor levels. For models using plant group as an explanatory variable, significant ANOVA results were followed by post-hoc pairwise tests (with P value adjustment for multiple testing; Benjamini and Hochberg 1995) to determine which levels of the factor generated the significant finding. For models with only continuous explanatory variables remaining after model simplification, t-tests were used to assess their significance.

4.3 Results

There was a significant relationship between ignition score and PC2 (χ^2 = 23.88, DF = 1, P < 0.0001). High-flammability species possessed shoots with comparatively low water content (Fig. 4.2). The trait PC1 (χ^2 = 0.19, DF = 1, P = 0.67) and the two interactions PC1 x status (χ^2 = 3.14, DF = 2, P = 0.21) and PC2 x status (χ^2 = 3.72, DF = 2, P = 0.16) were not significantly related to ignition score. Interspecific variation in BT was significantly related to PC1 ($F_{1,39}$ = 16.80, P = 0.0002) and PC2 ($F_{1,39}$ = 6.68, P = 0.01). High-flammability species were characterised by shoots with comparatively heavy shoot dry weight and low water content (Fig. 4.3). The two interactions PC1 x status ($F_{2,39}$ = 0.47, P = 0.63) and PC2 x status ($F_{2,39}$ = 2.51, DF = 2, P = 0.09) were not significantly related to BT. There was a significant relationship between CT and PC1 (F1,40 = 23.29, P < 0.0001) with high-flammability species distinguished by heavy shoot dry biomass (DB) (Fig. 4). The trait PC2 ($F_{1,40}$ = 1.64, P =0.21) and the two interactions PC1 x status ($F_{2,40}$ = 0.34, P = 0.72) and PC2 x status ($F_{2,40}$ = 0.44, P = 0.65) were not significantly related to CT. Interspecific variation in BC was significantly related to PC1 ($F_{1,39}$ = 5.03, P = 0.03) and PC2 ($F_{1,39}$ = 10.57, P = 0.002). High-flammability species were characterised by shoots with comparatively heavy shoot DB and low water content (Fig. 4.3). The two interactions PC1 x status ($F_{2,39}$ = 0.47, P = 0.63) and PC2 x status ($F_{2,39}$ = 2.51, DF = 2, P = 0.09) were not significantly related to BT.

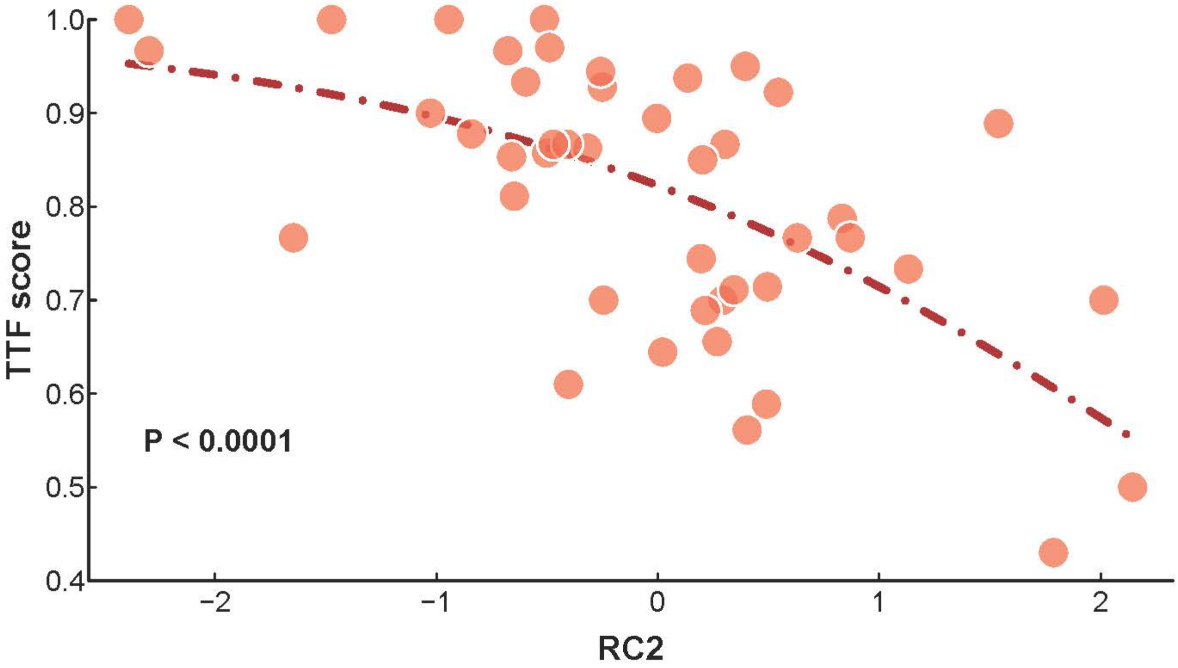

Fig. 4.2 ignition score (y-axis) as function of shoot water content (x-axis) from RC2. Data points are species' mean values. P = statistical significance. Dot-dash line represents line of best fit.

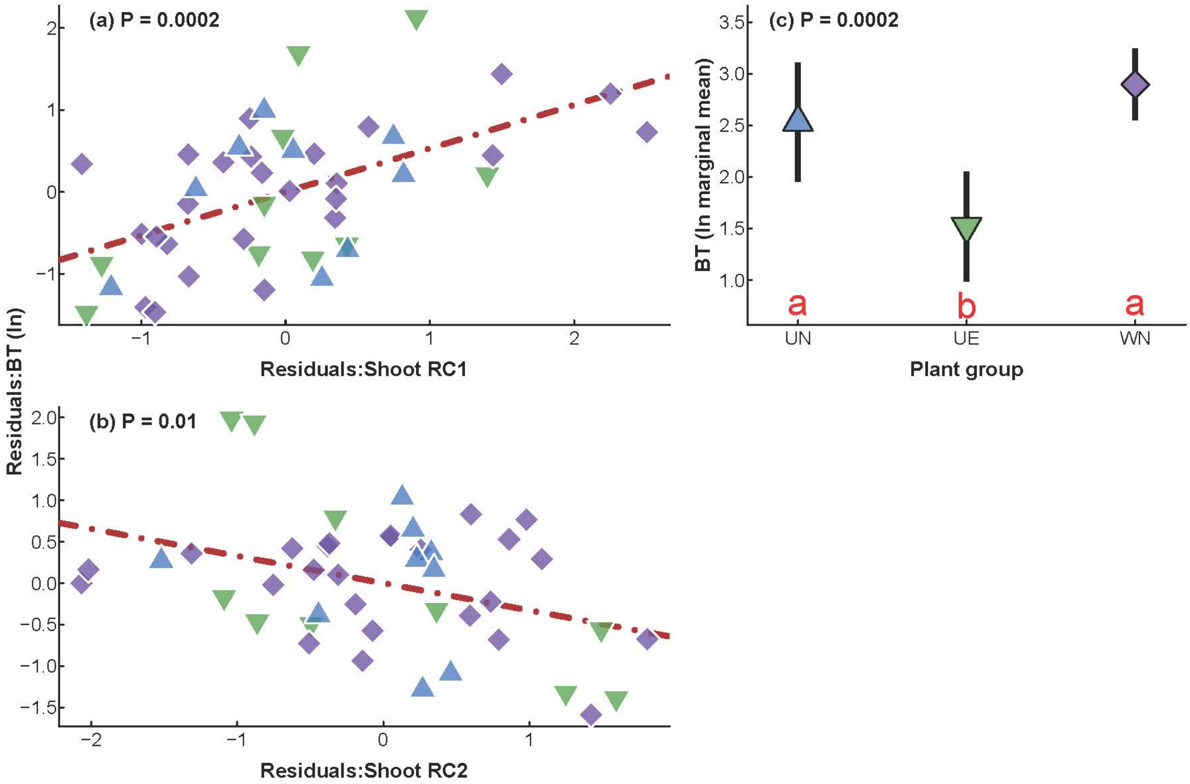

Fig. 4.3 (a) BT (y-axis) in relation to shoot RC1 (x-axis), which represents shoot dry weight. (b) BT (y-axis) in relation to shoot RC2 (x-axis), which represents shoot water content. (c) Mean BT (y-axis) for each plant group (x-axis). Each point on plots represents plant group's mean values, whiskers represent range in values across group. Triangles = urban species; diamonds = wildland species. Blue = urban native species; green = urban-exotic species; purple = wildland native species. P = statistical significance. Letters indicate direction of statistical significance.

I found that plant status was significantly related to BT (F2,39 = 10.38, P = 0.0002), CT (F2,40 = 4.15, P = 0.02) and BC (F2,39 = 4.41, P = 0.02), but not to ignition score (χ2 = 1.52, DF = 2, P = 0.47). Wildland native plants were more flammable than urban exotic plants for BT, CT and BC (Figs 4.3–4.5). In addition for BT, urban native plants were more flammable than urban exotic plants (Fig. 4.3). The significant and unique relationships between status and each of these three flammability attributes after model simplification indicates that plant traits not measured in this study, and that are associated with differences between native and exotic species, must be driving higher shoot flammability in native plants.

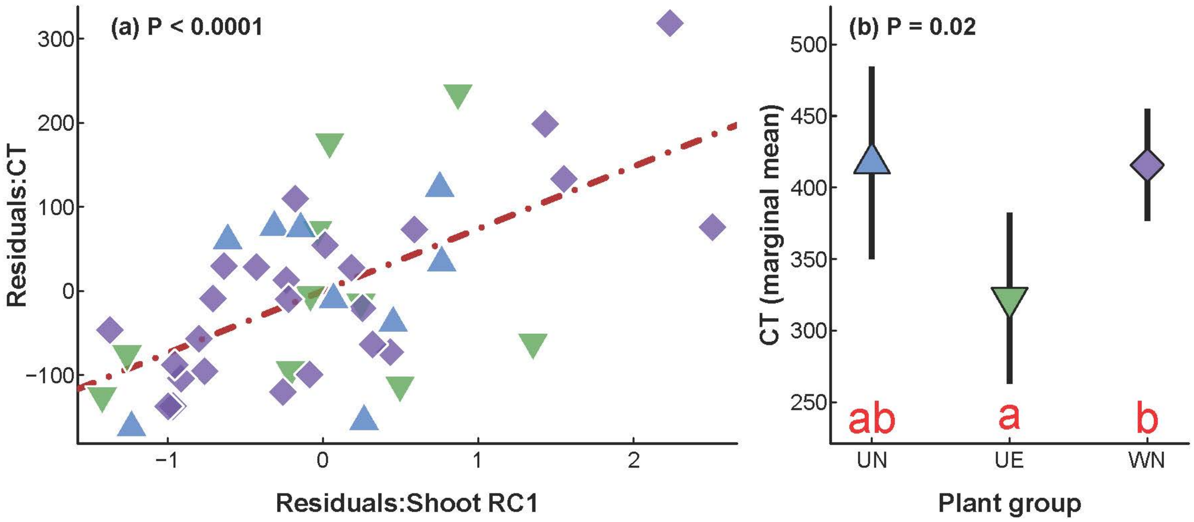

Fig 4.4 (a) Relationship between CT (y-axis) and shoot RC1 (x-axis), which represents shoot dry biomass (DB). Data points represent species mean values. Dot-dash line represents line of best fit. (b) CT (y-axis) as a function of plant group (x-axis). Data points represent plant group's mean values, whiskers represent range in values across group. Triangles = urban species; diamonds = wildland species. Blue = urban native species; green = urban-exotic species; purple = wildland native species. P = statistical significance. Letters indicate direction of statistical significance.

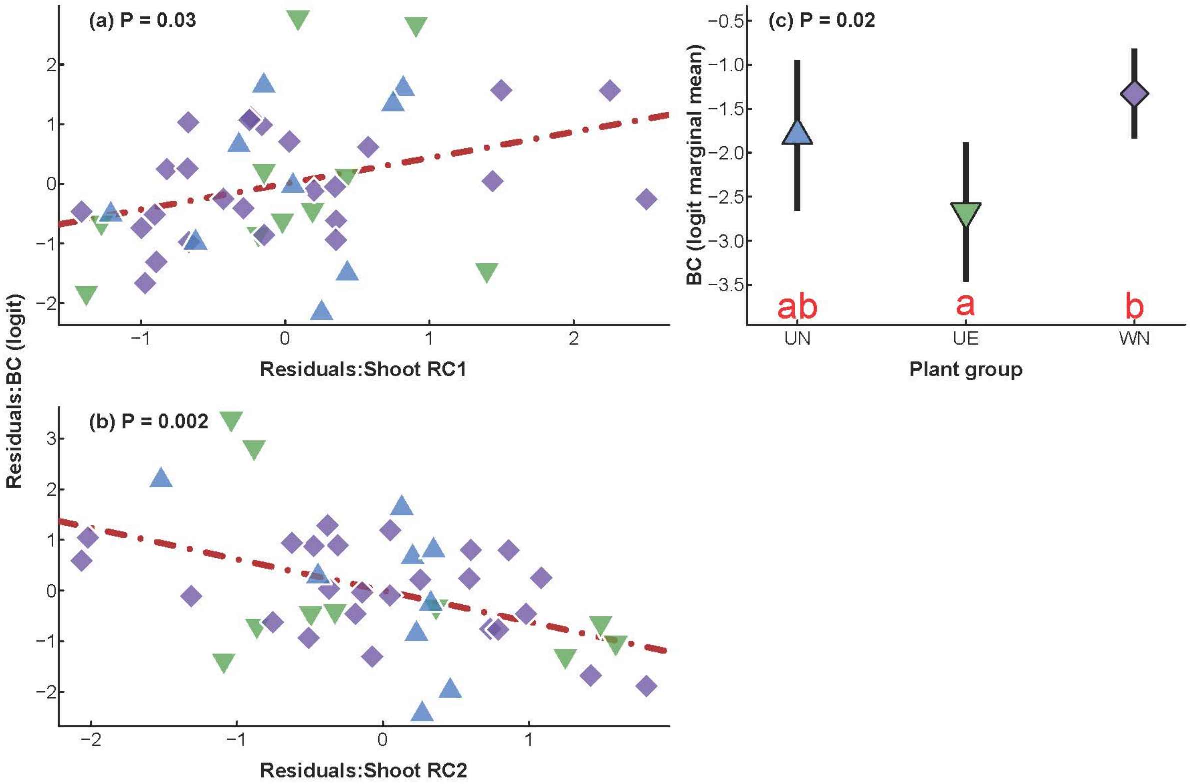

Fig. 4.5 (a) BC (x-axis) as a function of shoot RC1 (y-axis), which represents shoot water content. (b) BC (x-axis) as a function of shoot RC1 (y-axis), which represents shoot dry mass. Data points in (a) and (b) represent species mean values. Dot-dash line represents line of best fit. (c) BC (x-axis) as a function of plant group (y-axis). Each point on plots represents plant group's mean values, whiskers represent range in values across group. Triangles = urban species; diamonds = wildland species. Blue = urban native species; green = urban-exotic species; purple = wildland native species. P = statistical significance. Letters indicate direction of statistical significance.

The comparisons of PC1 ($F_{2,41}$ = 4.04, P = 0.03) and PC2 (F2,41 = 2.72, P = 0.08) among wildland native, urban native and urban exotic plants showed that only PC1 differed significantly among these plant groups (Fig. 4.6). However, the only significant pairwise comparison for PC1 was between wildland native and urban native plants, with urban native plants having higher values of PC1 (i.e. shoots with comparatively heavy shoot dry biomass) than wildland native plants (Fig. 6). The lack of significant differences in either PC1 or PC2 between native and exotic plants demonstrates that shoot dry biomass and shoot water content cannot explain differences in shoot flammability between native and exotic plants.

4.4 Discussion

There is a large range of interspecific shoot-level flammability reported in other biomes across the globe (Calitz *et al.* 2015; Wyse *et al.* 2016; Cubino *et al.* 2018; Santacruz-Garcia *et al.* 2019; Cui *et al.* 2020b). Similar variability was found in Chapter 3 for plant species in this fire-prone WUI. Therefore in this chapter, through measuring key shoot traits, I sought to understand interspecific differences in flammability across species from urban and wildland areas in this fire-prone WUI. I found that SWC and DB were significant shoot traits for explaining differences in ignitability (ignition score), sustainability (BT), combustibility (CT) and consumability (BC) of common plants at this WUI. While DR was reported as important shoot traits in other studies (Santacruz-Garcia *et al.* 2019), it was not an important trait in species from this study region. Additionally, BD was not a significant determinant of shoot flammability in this study which was consistent in other shoot flammability work (Cubino *et al.* 2018). In terms of shoot traits driving plant group (native-exotic, wildland-urban) differences in flammability, I found there must be other more important traits that influence differences between groups that were not measured in this study.

There was a significant negative relationship between SWC and shoot ignitability, sustainability and consumability. This relationship is rather intuitive where species that were highly flammable (higher ignition scores, longer BT and higher BC) possessed shoots with comparatively low SWC and low flammability species (low ignition scores, low BT, low BC) had high SWC. This pattern has been widely reported at the leaf scale (Alessio *et al.* 2008; Murray *et al.* 2013; Bianchi and Defosse 2015; Krix and Murray 2018) and emerging at the shoot flammability scale (Wyse *et al.* 2018; Cui *et al.* 2020a). Fuel dryness extends to landscape-scale fires as one of the four limiting factors of large wildfire occurrence (Bradstock 2010). Future climate conditions project that fire regimes in Australia may change, where dry woodland communities (biomass limited) may decrease in fire frequency and temperate forests (moisture dependent) may increase in fire frequency (Bradstock 2010). Severe drought in the 2019-20 fire season provided optimal conditions for large wildfires in temperate forest communities in NSW, including parts of the study region (Nolan *et al.* 2020). Whilst many of the temperate communities affected were resilient Eucalypt-dominant vegetation assemblages, typical moist refugia-type habitats, including fire-sensitive Gondwana Rainforest were severely burnt (Nolan *et al.* 2020). It is therefore of great importance that we understand shoot flammability relationships with fuel moisture content in a range of different vegetation communities as the climate changes.

Species in the study region that had heavy DB exhibited high flammability in terms of sustainability, consumability and combustibility. To my knowledge, shoot flammability papers seldom contain DB as a trait (Wyse *et al.* 2018), but clearly this trait is significant for understanding interspecific differences in flammability. This trait represents the available plant tissue to burn during combustion which is likely to represent a gross volume of cellulose, hemicelluloses and lignin as these structural compounds make up the bulk of woody plant tissue (Berry and Roderick 2005). At the leaf scale, higher concentrations of lignin and cellulose reduced the ability of Lodgepole pine foliage to sustain flaming combustion (Page *et al.* 2012). However, lignin was confounded by cellulose as the method that allowed quantifying these compounds did not allow researchers to establish the proportions of cellulose and lignin (Page *et al.* 2012). Conversely, Alam *et al.* (2019) found leaf lignin content to be the most important chemical trait affecting shoot flammability. Species that were high in leaf lignin content were had higher values of sustainability, consumability and combustibility (Alam *et al.* 2019). Whilst I was not able to quantify lignin and cellulose content my results show synonymous patterns with Alam *et al.* (2019), where shoot DB increased with sustainability, consumability and combustibility. Further work could look at quantifying lignin and cellulose contents at the leaf and shoot scales.

In chapter 3, I found that differences in flammability between urban and wildland groups were driven by the presence of low flammability exotic species in the urban areas of this WUI. Surprisingly, the two significant shoot traits in this chapter (SWC and DB) were not important in explaining differences between plant groups (wildland-native, urban-native, and urban-exotic). Plant traits not measured in this study, and that are associated with differences between native and exotic species, must be driving higher shoot flammability in native plants. Despite fuel moisture content being widely reported as an important trait influencing fuel flammability (Murray *et al.* 2013; Bianchi and Defosse 2015; Simpson *et al.* 2016; Wyse *et al.* 2018; Cui *et al.* 2020a), urban-exotic species were not simply lower in flammability due to higher SWC values than wildland and urban-native groups. Burger and Bond (2015) found fuel moisture content to be negligible when comparing plant groups with contrasting post-fire recruitment strategies in South Africa. Given that shoot traits are not adequate drivers of differences between plant groups in this study, I would like to suggest investigating individual leaves to understand differences between native and exotic species in this WUI system. Alam *et al.* (2019) showed that leaf morphological and chemical traits are important predictors of shoot flammability in native and exotic species in New Zealand. Volatile organic compounds (VOCs) such as terpenes and terpenoids within leaves are well studied and have been shown to increase leaf-level flammability (Owens *et al.* 1998; Alessio *et al.* 2008; Pausas *et al.* 2016;

Della Rocca *et al.* 2017). The surrounding vegetation of the WUI is important context for understanding difference between urban and wildland groups. The Greater Blue Mountains is dominated by Eucalypt-dominant vegetation assemblages which have had a strong evolutionary history with fire (Crisp *et al.* 2011). Considering the growing popularity for the Mutch (1970) hypothesis (Bond and Midgley 1995; Schwilk and Kerr 2002; Cowan and Ackerly 2010; Pausas *et al.* 2012; Burger and Bond 2015), there is a chance that many wildland plants here have evolved traits that may enhance their flammability. The presence of terpenes may be one of such traits (He and Lamont 2018). Among the 26 wildland species were genera *Acacia, Angophora, Callistemon, Corymbia, Leptospermum, Lophostemon, Syncarpia*, known to contain high concentrations of VOCs (Leach and Whiffin 1989; Brophy *et al.* 2000; Padovan *et al.* 2014; Avoseh *et al.* 2015). Therefore, there is a chance that high flammability native species may contain high concentrations of terpenes. I recommend that future studies quantify terpenes and other leaf chemical traits when comparing native and exotic species in Australian WUI systems to better understand differences between native and exotic species.

Chapter 5 | General Discussion

5.1 Were the aims of this thesis met?

In this thesis I sought to determine differences in floristic composition of dominant plants between urban and wildland vegetation assemblages at a fire-prone WUI and then to explore interspecific patterns in shoot flammability and shoot traits of key WUI species to better understand plant flammability at this WUI. My work in this thesis has met these aims by (1) showing significant floristic differences between key areas of the WUI, (2) identifying common and widespread plants at the WUI, (3) quantifying shoot flammability of common and widespread WUI plants, (4) comparing interspecific variation in shoot flammability of WUI plants, and (5) investigating shoot traits to better understand interspecific variation in shoot flammability and differences in shoot flammability between native and exotic species in the WUI.

5.1.1 Plant composition at the WUI

Across 110 sites, there were significant floristic differences in the dominant plant species across the four broad areas dry sclerophyll woodland (DS), wet sclerophyll gully (GU), homes and gardens (HG) and parks and recreational areas (PR) of the WUI. The most significant result in terms of understanding wildfire hazards and flammability at the WUI was that urban areas (in particular HG sites) had high exotic species richness and canopy cover combined with low native species richness and canopy cover. This pattern was generally repeated across vegetation strata (understorey, mid-storey and overstorey) but there were subtle differences (see Chapter 2). This is significant because exotic species have been shown to influence wildfire dynamics (Cubino *et al.* 2018) and considering they can be more flammable than native species (Murray *et al.* 2013; Wyse *et al.* 2016), they can enhance the flammability of environments they invade (Rossiter *et al.* 2003).

5.1.2 Comparisons of shoot flammability between urban and wildland areas at the WUI

I quantified the shoot flammability 45 woody plant species that were locally abundant in urban and wildland areas of this WUI. Wildland plants were more flammable than urban plants in terms of sustainability (BT) and consumability (BC). Similarly, native plants were more flammable than exotic plants in terms of BT, BC as well as combustibility (CT). Native and exotic plants occurred in both urban and wildland areas. Therefore, when I examined subset groups further, I found that wildland-native and urban-native plants were more flammable than urban-exotic plants in BT, BC and CT. Therefore, differences in shoot flammability between urban wildland groups is driven by the presence of low flammability exotic species in urban areas – a pattern matching the floristic differences established in Chapter 2. There is variation in the nature of native and exotic species' flammability reported and in some cases my findings are supported (Cubino *et al.* 2018) in others they are contrasted (Wyse *et al.* 2016). This suggests that the influence that exotic species have on flammability of an area is context-dependent. Ultimately these findings further highlight the fire-proneness of the surrounding wildland vegetation and the scale of work to be done on identifying low flammability, biodiversity promoting plants for reducing wildfire impacts at WUIs.

5.1.3 Shoot traits drive interspecific shoot flammability

I observed considerable interspecific variation in shoot flammability in Chapter 3. Given that shoot flammability preserves shoot traits (Jaureguiberry *et al.* 2011), I then investigated whether interspecific differences could be explained by shoot traits. I measured shoot water content (SWC), shoot dry biomass (DB), degree of ramification (DR), and bulk density (BD) as potential traits that underpin differences in shoot flammability. Through principle components I found that SWC and DB (68.2% of variation) were the most important shoot traits driving interspecific differences in ignitibility (ignition scores), BT, BC, and CT. Generally, species with low SWC had high ignition scores, long BTs and high % BC during burning. Plants that had high shoot DB were generally high flammability species with high BTs, high %BC and high CTs. There was no significant findings in terms of DR or BD.

Given there were plant group differences in flammability, I investigated if any shoot traits were important in explaining plant group (wildland-native, urban-native, and urban-exotic) differences. Through the use of minimum adequate models (MAMs) I found that plant group was significant, indicating that there were other unmeasured traits that were important in explaining differences between plant groups in flammability that was not measured here. Considering that leaf chemistry has been shown to underpin shoot flammability (Alam *et al.* 2019) and that volatile oils have been widely reported to increase flammability (Alessio *et al.* 2008), I propose that volatile oil content in native species could be driving differences between plant groups.

5.2 Future research directions

The work in this thesis provides the first analysis of shoot flammability for Australian vegetation. Within the broader context of fire ecology, shoot flammability has been employed in a range of research areas across the globe including understanding evolutionary adaptations to fire (Burger and Bond 2015; Calitz *et al.* 2015; Battersby *et al.* 2017a; Cui *et al.* 2020a; Cui *et al.* 2020b), determining invasive species effects on fire regimes (Cubino *et al.* 2018; Wyse *et al.* 2018) and understanding plant morphological trait relationships with flammability attributes (Wyse *et al.* 2016; Alam *et al.* 2019; Santacruz-Garcia *et al.* 2019). Therefore, the scope for future shoot flammability research in Australia is enormous. In this section, I provide some future directions to be explored relevant to my findings in this body of work.

One key finding from this research was that dominant wildland plants are more flammable than dominant urban plants at this WUI. This has serious implications for wildfire risk at this WUI. The wildland vegetation of the Blue Mountains comprises mostly of highly flammable eucalypt-dominant vegetation assemblages, however approximately 1% of the Greater Blue Mountains contains rainforest vegetation communities (Hammill and Tasker 2010). In the Greater Blue Mountains there are patches of the threatened ecological community (TEC), Western Sydney Dry Rainforest (OEH 2000). Rainforest communities are renowned for maintaining high moisture contents year round and have been referenced as "islands of green in a land of fire" here in Australia (Bowman 2000). Given that shoot water content was an important variable in explaining variation in shoot flammability and that low flammability species maintained high shoot water content (see Chapter 4), Western Sydney Dry Rainforest could provide a starting point to identifying low flammability native species. Murray *et al.* (2018) provided a framework where ideal green firebreak species could be selected from a pool of commercially available species based on five selection criteria. Species must be low in flammability, biodiversity promoting, weather proof, health safe and have community value (Murray *et al.* 2018). Future research could look start by screening native species from Western Sydney Dry Rainforest for low flammability attributes. After low flammability species are determined, the widespread planting of these species could not only reduce wildfire hazards in this fire-prone WUI but also enhance biodiversity and help regenerate this TEC in the Cumberland Plain.

Plants' evolutionary history with fire is well-researched where fire regimes have shaped plant communities across the globe (He *et al.* 2019). Mutch (1970) proposed that fire-prone plant communities are more flammable than fire-free plant communities because natural selection may have favoured the evolution of flammable traits. This hypothesis has been under much debate in the literature (Bond and Midgley 1995; Midgley 2013; Bowman *et al.* 2014) but appears to have grown in popularity. Studies in fire-prone biomes across the globe have investigated the theory with mixed results (Pausas *et al.* 2012; Clarke *et al.* 2014; Burger and Bond 2015). Interestingly, in Australia few studies have chosen to explore the idea (but see Clarke et al. 2014), despite Australian vegetation communities being among the most fire-prone in the world (Bradstock 2010). In Chapter 3, I identified three Proteaceous species that had high flammability attributes and were known to exhibit serotiny as a post-fire reproductive strategy (Pausas and Lamont 2018). This may suggest evidence to support the theory of Mutch (1970) could be expressed in fire-prone plant communities in Australia. Considering that Proteaceae in Australia is well-researched (Lamont *et al.* 1991; Enright *et al.* 1996; He *et al.* 2011; Pausas and Lamont 2018), further work that compares shoot flammability of Proteaceous species from fire-prone vegetation communities (eg. *Isopogon anemonifolius*) with those from fire-free communities (eg. *Stenocarpus sinuatus*) may provide interesting insights into evolution of flammability as a trait.

In this study, the four shoot traits (SWC, DB, DR, BD) measured were unable to explain differences in shoot flammability between native and exotic species (see Chapter 4). We were able to show that significant differences between native and exotic species must be underpinned by some other trait that was not measured here. Alam *et al.* (2019) showed that there are important leaf chemical and leaf morphological traits that drive interspecific variability in shoot flammability. The foliage of major Australian plant families have been shown to contain high amounts of volatile oils, in the form of terpenes (Leach and Whiffin 1989; Brophy *et al.* 2000; Avoseh *et al.* 2015). Plants that are high in terpenes have been shown to be high in flammability (Alessio *et al.* 2008). Considering that native species were generally more flammable than exotic species in this study, I suggest that future studies look at terpene concentrations in Australian native species. This approach may elucidate differences in shoot flammability between native and exotic species where shoot traits are inadequate.

5.3 Conclusion

This thesis took a novel approach to understanding wildfire hazards at the WUI. I showed that vegetation composition among WUI areas is different and that these differences are driven by the dominance exotic species in household gardens. For the first time, I used the well-regarded shoot flammability device to investigate flammability of common and widespread plants at the WUI. I showed that in this WUI, wildland plants are generally more flammable than urban plants and that this pattern is driven by the presence of exotic species in urban areas of the study region. This aspect of the study highlights the scale of flammability work still to be done to determine ideal species for green firebreaks in Australia and highlights the fire-prone nature of the vegetation surrounding this WUI. I also showed that some shoot traits are important for understanding why plants at the WUI differ in flammability. However, further work is required to determine why wildland plants are more than flammable urban plants in this WUI system. Investigating leaf chemical and morphological traits could provide the missing lens to understanding these differences. I also hope that the research in this thesis may be useful in improving the accuracy of wildfire models that incorporate plant traits and flammability as demonstrated in work by Zylstra *et al.* (2016).

Appendix

Supporting information for Chapter 2

Appendix 2.1 Plant species recorded across the study sites with taxonomic family, status (native = N, exotic = E) and growth form.

Species	Family	Status	Growth form
Acacia clunies-rossiae	Fabaceae	N	Tree
Acacia floribunda	Fabaceae	N	Shrub
Acacia linifolia	Fabaceae	N	Shrub
Acacia longifolia	Fabaceae	N	Shrub
Acacia mearnsii	Fabaceae	N	Shrub
Acacia obtusifolia	Fabaceae	N	Shrub
Acacia parvipinnula	Fabaceae	N	Shrub
Acacia terminalis	Fabaceae	N	Shrub
Acacia ulicifolia	Fabaceae	N	Shrub
Aechmea cylindrata	Bromeliaceae	E	Bromeliad
Agapanthus orientalis	Amaryllidaceae	E	Herb
Agave attenuata	Asparagaceae	E	Herb
Allocasaurina littoralis	Casuarinaceae	N	Tree
Allocasaurina torulosa	Casuarinaceae	N	Tree
Aloe arborescens	Asphodelaceae	E	Shrub
Angophora costata	Myrtaceae	N	Tree
Araucaria heterophylla	Araucariaceae	N	Tree
Asparagus aethiopicus	Asparagaceae	E	Herb
Banksia ericifolia	Proteaceae	N	Shrub
Banksia integrifolia	Proteaceae	N	Shrub
Banksia marginata	Proteaceae	N	Shrub
Banksia serrata	Proteaceae	N	Shrub
Banksia spinulosa	Proteaceae	N	Shrub
Bossiaea obcordata	Fabaceae	N	Shrub
Bossiaea rhombifolia	Fabaceae	N	Shrub
Buxus microphylla	Buxaceae	E	Shrub
Callicoma serratifolia	Cunoniaceae	E	Tree

Species	**Family**	**Status**	**Growth form**
Callistemon viminalis	Myrtaceae	E	Shrub
Calochlaena dubia	Dicksoniaceae	E	Fern
Camellia japonica	Theaceae	E	Shrub
Camellia sasanqua	Theaceae	E	Shrub
Citrus limon	Rutaceae	E	Tree
Cycas revoluta	Cycadaceae	E	Cycad
Corymbia eximia	Myrtaceae	N	Tree
Corymbia gummifera	Myrtaceae	N	Tree
Doryanthes excelsa	Doryanthaceae	N	Lily
Eucalyptus capitellata	Myrtaceae	N	Tree
Eucalyptus crebra	Myrtaceae	N	Tree
Eucalyptus deanei	Myrtaceae	N	Tree
Eucalyptus eugenioides	Myrtaceae	N	Tree
Eucalyptus oblonga	Myrtaceae	N	Tree
Eucalyptus parramattensis	Myrtaceae	N	Tree
Eucalyptus piperita	Myrtaceae	N	Tree
Eucalyptus punctata	Myrtaceae	N	Tree
Eucalyptus resinifera	Myrtaceae	N	Tree
Eucalyptus saligna	Myrtaceae	N	Tree
Eucalyptus sclerophylla	Myrtaceae	N	Tree
Grevillea banksii x *bipinnatifida*	Proteaceae	N	Shrub
Grevillea banksii	Proteaceae	N	Shrub
Grevillea phylicoides	Proteaceae	N	Shrub
Hakea bakeriana	Proteaceae	N	Shrub
Hakea dactyloides	Proteaceae	N	Shrub
Hakea propinqua	Proteaceae	N	Shrub
Hakea salicifolia	Proteaceae	N	Shrub
Hakea sericea	Proteaceae	N	Shrub
Jacaranda mimosifolia	Bignoniaceae	E	Tree
Lambertia formosa	Proteaceae	N	Shrub
Lantana camara	Verbenaceae	E	Shrub
Lavandula angustifolia	Lamiaceae	E	Herb
Leptospermum juniperinum	Myrtaceae	N	Shrub

Species	Family	Status	Growth form
Leptospermum trinervium	Myrtaceae	N	Shrub
Liquidambar styraciflua	Altingiaceae	E	Tree
Lomandra longifolia	Lomandraceae	N	Herb
Lophostemon confertus	Myrtaceae	N	Tree
Magnolia grandiflora	Magnoliaceae	E	Tree
Magnolia grandiflora 'Little Gem'	Magnoliaceae	E	Shrub
Melaleuca quinquenervia	Myrtaceae	N	Tree
Monstera deliciosa	Araceae	E	Climber
Murraya paniculata	Rutaceae	E	Shrub
Nephrolepis exaltata	Lomariopsidaceae	E	Fern
Olea europaea	Oleaceae	E	Tree
Osteospermum fruticosum	Asteraceae	E	Herb
Panicum simile	Poaceae	N	Grass
Pelargonium citrosum	Geraniaceae	E	Herb
Petrophile pulchella	Proteaceae	N	Shrub
Photinia x *fraseri* 'Red Robin'	Rosaceae	E	Shrub
Pittosporum undulatum	Pittosporaceae	N	Tree
Pittosporum tenuifolium 'Silver Sheen'	Pittosporaceae	E	Shrub
Plumeria alba	Apocynaceae	E	Tree
Poa labillardieri	Poaceae	N	Grass
Prunus serrulata	Rosaceae	E	Tree
Pultenaea linophylla	Fabaceae	N	Shrub
Pultenaea scabra	Fabaceae	N	Shrub
Quercus robur	Fagaceae	E	Tree
Rhododendron 'White Pearl'	Ericaceae	E	Shrub
Rhododendron 'Sneezy'	Ericaceae	E	Shrub
Rosmarinus officinalis	Lamiaceae	E	Herb
Strelitzia reginae	Strelitziaceae	E	Herb
Syncarpia glomulifera	Myrtaceae	E	Tree
Trachelospermum jasminoides	Apocynaceae	E	Climber
Tropaeolum majus	Tropaeolaceae	E	Climber
Cryptomeria japonica 'Elegans'	Cupressaceae	E	Climber
Ulmus parvifolia	Ulmaceae	E	Tree

Species	**Family**	**Status**	**Growth form**
Ulmus procera	Ulmaceae	E	Tree
Patersonia fragilis	Iridaceae	N	Herb
Imperata cylindrica 'Rubra'	Poaceae	E	Grass
Syagrus romanzoffiana	Arecaceae	E	Tree
Pinus radiata	Pinaceae	E	Tree
Viburnum odoratissium	Adoxaceae	E	Shrub
Woollsia pungens	Ericaceae	N	Shrub
Yucca filamentosa	Asparagaceae	E	Shrub
Xanthorrhoea minor	Xanthorrhoeaceae	N	Grass

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