Predicting fire in rainforest

by

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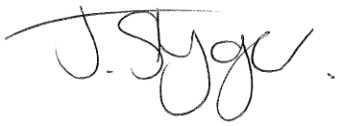

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Abstract

Cool-temperate rainforest occurs widely within south-west and western Tasmania, where it occurs interspersed with buttongrass moorlands. Rainforest is considered to be a climax vegetation, capable of regenerating in the absence of a major disturbance event, such as fire. Rainforest is also considered to be a fire sensitive community, as many rainforest species are incapable of surviving a fire event. Although fire in rainforest is rare, large rainforest fires have occurred in the past. These fire events are likely to increase with future climate change, which may result in a substantial loss of rainforest communities. It is important to understand the conditions under which fire will sustain and spread within rainforest as this will aid in protective measures, such as hazard-reduction burning, and the allocation of resources during a wildfire.

In this study, I ask, under what conditions it would be likely that a fire would sustain and spread within rainforest. In order to do this the flammability and microclimate of a callidendrous rainforest, implicate rainforest and deciduous beech montane rainforest were characterised. The canopy structure and rainfall distribution of the callidendrous rainforest were also examined. There was very little difference in the flammability of live leaf and litter components between the three rainforest communities and adjacent fire tolerant communities, with the exception of the bark component from a *Eucalyptus coccifera* woodland. Callidendrous and implicate rainforests were cooler, more humid and less windy than adjacent open areas. There was very little difference in temperature and vapour pressure deficit between the deciduous beech forest and the adjacent open area. The distribution of rainfall within a callidendrous rainforest was found to be heterogeneous. Two millimetres of rain was required to saturate the rainforest canopy. On average, 20% of rainfall was intercepted.

The Soil Dryness Index (SDI) is a tool used by fire managers to provide an indication of drought conditions and is also a component of the McArthur Forest Fire Danger Index (FFDI) in Tasmania. Many fire managers believe that the SDI does not perform effectively in south-west and western Tasmania. As a result, the performance of the SDI was looked at in this region, by examining the canopy intercept factor used to calculate rainforest and the relative performance of the SDI between mineral and organic soils. It was found that the canopy intercept factor designated for rainforest within the SDI performed well, and the SDI for rainforest could not be improved by using the canopy intercept rule determined for callidendrous rainforest earlier in this study. It was also found that there was no difference in the way the SDI performed between mineral and organic soils. It was therefore thought that the observed poor performance of the

SDI in south-west and western Tasmania is likely to be the result of a poor representation of weather stations in a topographically complex environment.

Twelve historical fires that either burned into, or stopped at rainforest boundaries, were examined. The rainfall in the past 10, 20, 30, 60, 100 and 365 days, as well as SDI, Drought Factor, temperature, relative humidity, wind speed and FFDI were determined for each fire to establish the best predictor of rainforest fire. It was found that the drought related variables were more important in predicting rainforest fire than the weather variables, with rainfall in the last 30 days above or below 50 mm being the most significant predictor of rainforest fire.

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Table of Contents

Declaration.....ii

Authority of access.....ii

Abstract.....iii

Acknowledgements.....v

List of Tables.....xii

List of Figures.....xiv

List of Plates.....xvii

Chapter 1 – Introduction.....1

1.1 Background.....1

1.2 Rainforest distribution.....2

1.3 Definition of Australian rainforest.....3

1.3.1 Tasmanian cool-temperate rainforest.....4

1.3.2 Mixed forest.....4

1.4 Rainforest and fire in Tasmania.....5

1.4.1 Mitigating rainforest fires – hazard reduction burning.....7

1.5 Fire danger indices.....8

1.5.1 McArthur Forest Fire Danger Index.....8

1.5.1.1 Soil Dryness Index.....8

1.5.1.2 Drought Factor.....9

1.5.2 Canadian Forest Fire Weather Index.....10

1.6 Thesis aims.....11

1.7 Structure of thesis.....12

Chapter 2 – Study sites.....13

2.1 Site selection and establishment.....13

2.1.1 Site Call....15
2.1.2 Site Imp....15
2.1.3 Site DB....16
2.1.4 Site Org....16
2.1.5 Control 1....17
2.1.6 Control 2....17
2.1.7 Control 3....18
2.2 Measurements of fuel moisture....18
2.3 Automatic weather station measurements....18
Chapter 3 – Flammability of rainforest components....20
3.1 Introduction....20
3.2 Methods....22
3.2.1 Study sites, site establishment and sampling procedure....22
3.2.2 Laboratory flammability experiments....24
3.2.2.1 Laboratory procedures....24
3.2.2.2 Flammability....25
3.2.2.3 Moisture content....26
3.2.3 Field flammability experiments....27
3.2.3.1 Test fire procedure....27
3.2.3.2 Weather data collection....27
3.2.4 Data analysis....28
3.2.4.1 Laboratory flammability experiments....28
3.2.4.2 Field flammability experiments....31
3.3 Results....31
3.3.1 Leaf samples....31
3.3.2 Woody litter samples....39
3.3.3 Vegetation type....43

3.3.4 Field test fires....44

3.4 Discussion....44

Chapter 4 – Structure of the rainforest canopy....50

4.1 Introduction....50

4.2 Methods....54

4.2.1 Site selection....54

4.2.2 Distribution of rain gauges....54

4.2.3 Data collection....54

4.2.4 Data analysis....55

4.3 Results....58

4.3.1 Temporal variation....60

4.3.2 Spatial analysis....65

4.3.3 Canopy saturation and interception....71

4.4 Discussion....74

Chapter 5 – Predicting rainforest microclimate....81

5.1 Introduction....81

5.1.1 Rainforest microclimate....81

5.2 Methods....82

5.3 Results....83

5.3.1 Air temperature....85

5.3.2 Vapour pressure deficit....88

5.3.3 Wind speed and direction....91

5.4 Discussion....92

Chapter 6 – Soil Dryness Index....98

6.1 Introduction....98

6.2 Methods....99

6.3 Results....101

6.4 Discussion....106

Chapter 7 – Historical rainforest fires....111

7.1 Introduction....111

7.2 Methods....111

7.2.1 Calculation of the Forest Fire Danger Index....111

7.2.2 Rainforest mapping....112

7.2.3 Data analysis....112

7.3 Results and Discussion....112

7.3.1 Individual fires....114

7.3.1.1 1933/34 fire....114

7.3.1.2 Savage River fire....114

7.3.1.3 Harrison's Opening fire....115

7.3.1.4 Cape Sorell fire....115

7.3.1.5 Mount Frankland-Donaldson fire....115

7.3.1.6 Mount Castor fire....116

7.3.1.7 Cracroft River fire....116

7.3.1.8 Reynolds Creek fire....116

7.3.1.9 Heemskirk Road fire....117

7.3.1.10 Kilmore East – Murrindindi fire....117

7.3.1.11 Lake Mackintosh fire....118

7.3.1.12 Giblin River fire....120

7.3.2 Integration of historical fire data....121

Chapter 8 – Discussion and conclusions....125

8.1 Introduction....125

8.1.1 Community flammability....125

8.1.2 Predictors of rainforest fire....126

8.2 Other outcomes....127

8.2.1 Further study.....129

References.....130

Appendix 1 – Input variables for calculating the FFDI for each fire.....143

Appendix 2 – Rainfall values for different time periods preceding each fire..... 144

List of Tables

Table 2.1 Site variables....14

Table 2.2 Automatic weather station variables measured and instruments used for each site....19

Table 3.1 Species examined for the flammability experiments and their collection location....22

Table 3.2 Beaufort scale for estimating wind velocity....28

Table 3.3 Community scores for all leaf species examined....29

Table 3.4 Tukey's multiple range comparison test for species against residuals of rate of spread and moisture content....32

Table 3.5 General linear model for response variables to rate of spread (mm s^{-1}) for foliar leaf species....33

Table 3.6 Regression equations, *P*-values and *R*-sq for the relationship between rate of spread (mm s^{-1}) and percentage moisture loss of leaf species....35

Table 3.7 Surface area to volume ratios of foliar leaf species....36

Table 3.8 Pearson's product moment correlation coefficients for surface area to volume ratio and rate of spread (mm s^{-1}) for the six foliar species across all drying times....36

Table 3.9 Regression equations, *P*-values and *R*-sq for the relationship between time to consumption and moisture content of leaf species....37

Table 3.10 General linear model for response variables to time to ignition for all foliar leaf species....38

Table 3.11 Mean values and results of one-way ANOVA for litter size classes and rate of spread (mm s^{-1}) between different vegetation types....39

Table 3.12 Mean values and results of one-way ANOVA for rate of spread (mm s^{-1}) for litter fuel components in each vegetation type....39

Table 3.13 Regression equations, *P*-values and *R*-sq for the relationship between rate of spread (mm s^{-1}) and surface area to volume ratio of litter samples in all size classes....40

Table 3.14 Regression equations, *P*-values and *R*-sq for the relationship between time to sustained ignition and dead fuel moisture content for the litter samples....41

Table 4.1 Abbreviations associated with canopy interception that occur in Chapter 4....51

Table 4.2 Rainfall event number, start date of each event, gross rainfall, event type, canopy status at the start of each event, the duration of each event in hours, rainfall rate, mean wind direction, mean wind speed, wind speed range, throughfall, throughfall as a percentage of gross rainfall, interception, and interception as a percentage of gross precipitation....59

Table 4.3 Spatial variation of throughfall and canopy characteristics for the funnel rain gauges.....66

Table 5.1 Pearson's product moment correlation coefficients between Δ of each variable inside and outside the forests and the variable at the control.....84

Table 6.1 Altered evapo-transpiration values (mm/day). Mount values multiplied by 0.5.....100

Table 6.2 Altered evapo-transpiration values (mm/day). Mount values multiplied by 1.25.....100

Table 6.3 Name of weather station and its altitude used to record the Soil Dryness Index within south-west Tasmania, areas bordering south-west Tasmania, and the west coast of Tasmania.....109

Table 7.1 Summary table of fires and Forest Fire Danger Index.....113

Table 7.2 Percentage rainforest burned, rainfall in the previous 30 days preceding the fire, rainfall in the previous 60 days preceding the fire and Soil Dryness Index (SDI).....121

Table 7.3 Results of Wilcoxon rank sum tests for determinants of successful rainforest fires.....124

Table 8.1 Soil Dryness Index guidelines for hazard-reduction burning in different vegetation types.....127

List of Figures

Figure 1.1 Distribution of rainforest within Australia.....2

Figure 1.2 Structure of the Canadian Forest Fire Weather Index System.....11

Figure 2.1 Location of study sites within Tasmania and nearby localities.....13

Figure 3.1 Interval plot of residuals from the regression of rate of spread (mm s^{-1}) on moisture content for each species.....32

Figure 3.2 Relationship between rate of spread (mm s^{-1}) and air dry moisture loss.....34

Figure 3.3 Interval plot for surface area to volume ratio of foliar leaf species.....35

Figure 3.4 Relationship between time to consumption and moisture content of leaves of selected species.....37

Figure 3.5 Relationship between time to ignition of leaves of selected species and hours air drying.....38

Figure 3.6 Relationship between time to sustained ignition of litter material and different litter moisture contents.....41

Figure 3.7 Interval plot of time to sustained ignition of litter and litter size class for callidendrous forest litter across all moisture contents.....42

Figure 3.8 Interval plot of time to sustained ignition of litter and litter size class for deciduous beech scrub litter across all moisture contents.....42

Figure 3.9 Interval plot of time to sustained ignition of litter and litter size class for *Eucalyptus coccifera* woodland litter across all moisture contents.....43

Figure 4.1 Size of rainfall events and sub-events.....60

Figure 4.2 Rate of rainfall (mm/hr) for each of the 11 rainfall events.....61

Figure 4.3 Rate of rainfall (mm/hr) of events and sub-events.....61

Figure 4.4 Total and cumulative gross rainfall and throughfall for separate events over time.....63-64

Figure 4.5 Hemispherical canopy images taken at the site of each funnel rain gauge for gauges 1-12.....67

Figure 4.6 Hemispherical canopy images taken at the site of each funnel rain gauge for gauges 13-25.....68

Figure 4.7 Map of the 25 rain gauges at Site Call with the location, species and diameter at breast height in cm of all trees and tall shrubs in a 5 m radius of each rain gauge.....70

Figure 4.8 Relationship between interception as a percentage of gross rainfall and gross rainfall for all events and sub-events....71

Figure 4.9 Gross rainfall and throughfall for all events and sub-events....72

Figure 4.10 Gross rainfall and interception for all events and sub-events....73

Figure 4.11 Relationship between interception as a percentage of gross rainfall and rainfall rate....74

Figure 5.1 Relationship between Δ for each variable and the corresponding variable at the control....85

Figure 5.2 Frequency histogram of the Δ between inside and outside forest conditions for all temperatures greater than 25 °C and relative humidities less than 25% at Control 1, for Site Call and Control 1....86

Figure 5.3 Frequecny histogram of the Δ between inside and outside forest conditions for all temperatures (°C) at Site Call and Control 1....86

Figure 5.4 Frequency histogram of the Δ between inside and outside forest conditions for all temperatures (°C) at Site Call and Control 1 when the total solar radiation is greater than 1.5 MJ/m² at Control 1....86

Figure 5.5 Frequency histogram of the Δ between inside and outside forest conditions for all temperatures (°C) at Site Imp and Control 1....87

Figure 5.6 Frequency histogram of the Δ between inside and outside forest conditions for all temperatures (°C) at Site Imp and Control 1 when the total solar radiation is greater than 1.5 MJ/m² at Control 1....87

Figure 5.7 Frequency histogram of the Δ between inside and outside forest conditions for all temperatures (°C) at Site DB and Control 2....87

Figure 5.8 Frequency histogram of the Δ between inside and outside forest conditions for all temperatures (°C) at Site DB and Control 2 when the total solar radiation is greater than 1.5 MJ/m² at Control 2....87

Figure 5.9 Frequency histrogram of the Δ between inside and outside forest conditions for all temperatures (°C) from 1/12/2010 to 28/2/2011 for Site DB and Control 2....88

Figure 5.10 Frequency histogram of the Δ between inside and outside forest conditions for vapour pressure deficit (kPa) when temperatures are greater than 25 °C and relative humidities less than 25% at control 1, for Site Call and Control 1....89

Figure 5.11 Frequency histogram of the Δ between inside and outside forest conditions for vapour pressure deficit (kPa) at Site Call and Control 1....89

Figure 5.12 Frequency histogram of the Δ between inside and outside forest conditions for vapour pressure deficit (kPa) at Site Imp and Control 1 when the total solar radiation is greater than 1.5 MJ/m² at Control 1....89

Figure 5.13 Frequency histogram of the Δ between inside and outside forest conditions for vapour pressure deficit (kPa) at Site Imp and Control 1.....90

Figure 5.14 Frequency histogram of the Δ between inside and outside forest conditions for vapour pressure deficit (kPa) when the total solar radiation is greater than 1.5 MJ/m² at Control 1.....90

Figure 5.15 Frequency histogram of the Δ between inside and outside forest conditions for vapour pressure deficit (kPa) at Site DB and control 2.....90

Figure 5.16 Frequency histogram of the Δ between inside and outside forest conditions for vapour pressure deficit (kPa) at Site DB and Control 2 when the total solar radiation is greater than 1.5 MJ/m² at Control 2.....90

Figure 5.17 Frequency histogram of the Δ between inside and outside forest conditions for vapour pressure deficit (°C) from 1/12/2010 to 28/2/2011 for Site DB and Control 2.....91

Figure 5.18 Frequency histogram of the Δ between inside and outside forest conditions for wind speed (m/s) at Site Call and Control 1.....91

Figure 5.19 Frequency histogram of the Δ between inside and outside forest conditions for wind speeds (m/s) at Site DB and Control 2.....92

Figure 5.20 Frequency histogram of the Δ between inside and outside forest conditions for wind speeds (m/s) from 1/12/2010 to 28/2/2011 for Site DB and Control 2.....92

Figure 6.1 Site Call average daily volumetric soil moisture and 24 hour rainfall (mm) until 9.00 am between 24 March and 27 May 2012.....102

Figure 6.2 Site Org average daily volumetric soil moisture and 24 hour rainfall (mm) until 9.00 am for Mount Read between 22 February and 24 April 2011.....102

Figure 6.3 Average daily volumetric soil moisture and daily Soil Dryness Index for Site Call between 31 January and 27 May 2012.....103

Figure 6.4 Average daily volumetric soil moisture and daily Soil Dryness Index for Site Org between 22 February and 24 April 2011.....103

Figure 6.5 Relationship between average daily volumetric soil moisture at Site Call and 9.00 am rainfall at Control 1.....104

Figure 6.6 Relationship between average daily volumetric soil moisture at Site Org and 9.00 am rainfall at Mount Read.....104

Figure 6.7 Comparison of the Soil Drynee Index calculated for interception class B and F for Scotts Peak Dam automatic weather station from 1 January to 28 February 2010.....105

List of Plates

Plate 5.1 Automatic weather station at Site DB in winter.....94

Plate 7.1 Lake Mackintosh, showing burned stags of dead trees protruding from the lake.....118

Plate 7.2 Rainforest burned on the edge of Lake Mackintosh on 31 January 2010.....119

Plate 7.3 Rainforest understorey burned by the Lake Mackintosh fire on January 2010.....119

Chapter 1 – Introduction

1.1 Background

Rainforest is an overarching term that generally refers to the lush, cool and shady forests that are so obviously different from the eucalypt forests and woodlands that dominate Australia. Rainforest communities occur along the eastern seaboard of Australia, with small patches also occurring in the Northern Territory and Kimberly region of Western Australia (Adam 1992; Bowman 2000; Lynch and Neldner 2000). Within Tasmania, rainforest occurs mainly in the west coast and south-west regions of the Island (Jarman and Brown 1983). Rainforest communities are composed of shade-adapted species that are capable of regeneration in the absence of disturbance (Jarman and Brown 1983; Lynch and Neldner 2000). Rainforest species generally show an intolerance of fire, particularly when compared to their eucalypt neighbours (Jackson 1968; Read 2005; Wood *et al.* 2011; Little *et al.* 2012). For this reason, rainforest communities are considered to be fire-sensitive (Brown and Podger 1982; Bowman and Brown 1986; Cullen and Kirkpatrick 1988) with fire resulting in either a loss of integrity of the community or its complete destruction (Hill and Read 1984; Shearman *et al.* 2009). However, the occurrence of fire in rainforest is rare. This has been thought to be due to the low flammability of foliage, a humid microclimate and a low fuel load (Jackson 1968), as well as the propensity for rainforest to occur in topographic fire refugia (Wood *et al.* 2011). Nevertheless, under extreme weather conditions of high temperature, low humidity and high soil dryness rainforest will occasionally burn (Hill 1982; Barker 1991; Marsden-Smedley 1998; Cruz *et al.* 2012).

Global climate change is predicted to result in an average warming of the earth's surface and ocean temperatures. In Tasmania, climate change modelling predicts an increase in temperatures across the State. Although the total annual rainfall is unlikely to change across Tasmania, there is predicted to be a change in regional patterning and seasonality, with a decrease in summer rainfall predicted for the west coast of Tasmania. Combined with an increase in pan evaporation over the summer months, the west coast of Tasmania is predicted to become drier (Grose *et al.* 2010). These drier conditions are likely to increase the frequency and severity of bushfires occurring in western Tasmania, with the probable result of more fires penetrating into rainforest.

Currently the conditions under which rainforest will sustain fire are unknown. Detailed studies have determined fire behaviour and sustainability for buttongrass moorlands (Marsden-Smedley 1998), grasslands (Leonard 2009) and dry eucalypt forest (Luke and McArthur 1978; Gould *et al.* 2007) but no such model exists for rainforest. Given the predictions of climate change for western Tasmania, an understanding of the fire potential of rainforest is needed.

1.2 Rainforest distribution

Within Australia, communities known as rainforest extend from the cool-temperate zone of Tasmania at 44° S, to the very tip of the Cape York Peninsula in the tropics at 10° S (Adam 1992; Bowman 2000; Lynch and Neldner 2000). There are also small patches extending into the Northern Territory and northern Western Australia (Figure 1.1; Adam 1992; Bowman 2000).

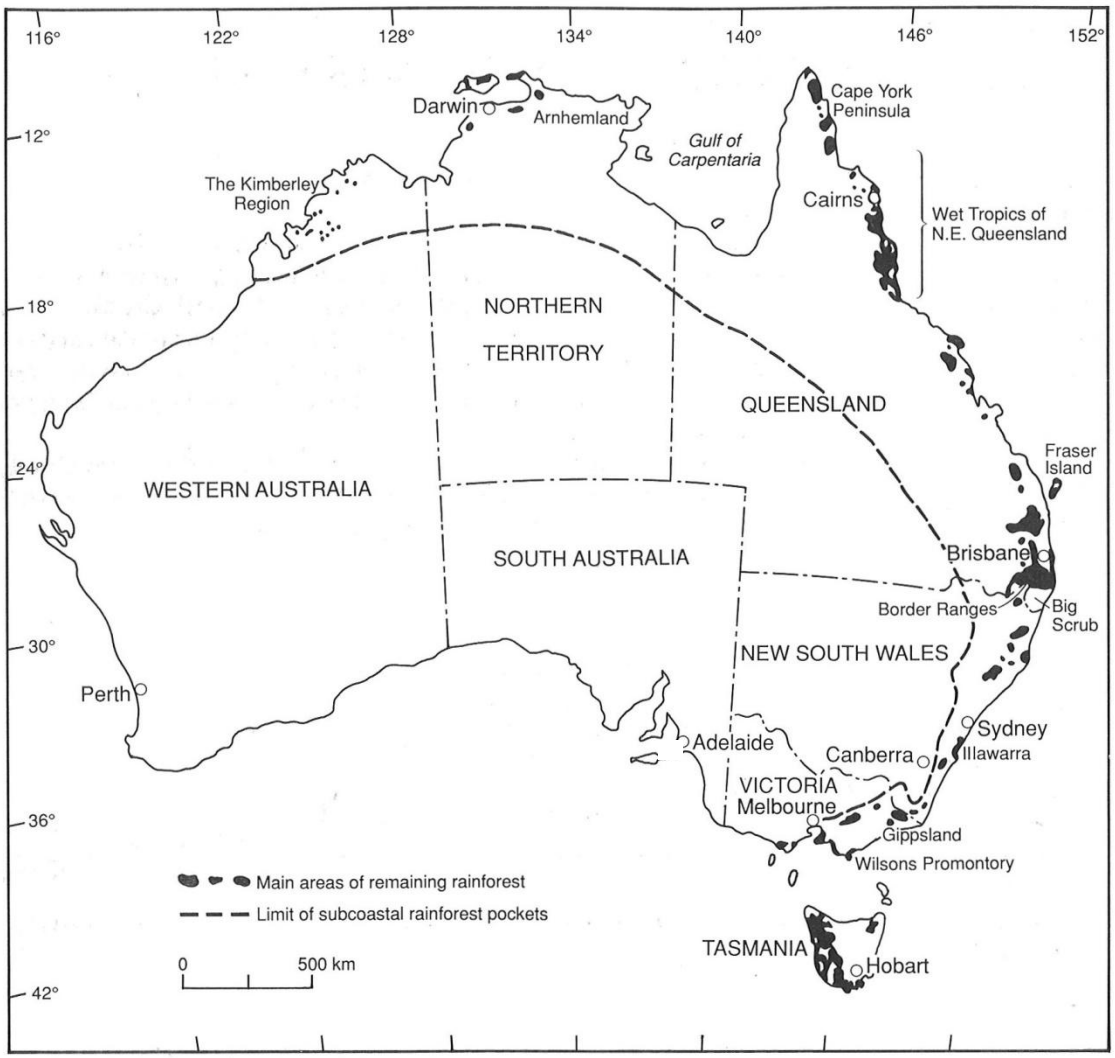

Figure 1.1 Distribution of rainforest within Australia (from Adam 1992).

Rainforest in Australia can occur in regions of annual rainfall varying from 600 mm to 3,600 mm, and on soils varying in fertility from high to extremely low (Bowman 2000). Rainforest also has a wide altitudinal range, occurring from sea level to the altitudinal limit of forest vegetation (Bowman 2000). However, its distribution within this potential range is highly restricted (Figure 1.1).

1.3 Definition of Australian rainforest

The definition of rainforest as a community has proved difficult (Jarman and Brown 1983; Lynch and Neldner 2000; Bowman 2000) and is often avoided altogether (Adam 1992). This difficulty in definition is partly due to different national and regional perceptions (Adam 1992). Herein, definitions of rainforest will be limited to those that have been developed for Australian forests, as well as some definitions that are applicable only to communities of rainforest occurring in Tasmania. However, even within Australia, the definition of rainforest can be problematic.

A number of complex taxonomic systems have been developed in attempt to classify Australian rainforest types however little agreement between authors and regions has been reached (Bowman 2000). Bowman (2000) lists a number of classificatory systems that can be used to define rainforest. These include, rainforest defined by climatological parameters; rainforest defined by *a priori* description; rainforest defined by diagnostic life forms; rainforest defined by forest floor light environment; rainforest defined by biogeographically distinct taxa; rainforest defined by fire susceptibility; and rainforest defined by what it is not (Bowman 2000).

Beadle and Costin (1952) proposed a classificatory system for Australian vegetation based on floristics and structure. Within this, they recognised rainforest as a single structural formation that could be divided into four subformations (Adam 1992). These subformations can be differentiated spatially and are known as tropical, subtropical, monsoon and temperate rainforests. Tropical and subtropical rainforests occur in northern Australia along the eastern seaboard, occupying wetter coastal areas and extend as far south as New South Wales (Lynch and Neldner 2000). Monsoon rainforests occur in northern and north-western Australia, in areas of seasonal dryness (Lynch and Neldner 2000). Temperate rainforests occur in eastern and south-eastern Australia and have been divided into two groups; warm-temperate and cool-temperate rainforests. Cool-temperate rainforest occurs from Victoria southward with a few

outliers occurring at high altitude sites in New South Wales and Queensland (Lynch and Neldner 2000).

1.3.1 Tasmanian cool-temperate rainforest

To overcome problems of a national rainforest definition many researchers have chosen to define and classify rainforest on a local scale (e.g. Webb (1959) in Queensland; Jarman and Brown (1983) in Tasmania; Floyd (1990) in New South Wales; and Russell-Smith (1991) in the Northern Territory). In Tasmania, cool-temperate rainforest has been defined using floristic composition and regeneration processes (Jarman and Brown 1983). This definition includes those communities dominated by trees of *Nothofagus*, *Atherosperma*, *Eucryphia*, *Athrotaxis*, *Lagarostrobos*, *Phyllocladus* or *Diselma* and able to regenerate, either vegetatively or from seed, in the absence of large-scale disturbance, such as fire. Tasmanian rainforest has been further classified into four forest types; callidendrous, thamnic, implicate and open montane rainforest (Jarman *et al.* 2005). The former three forest types intergrade in terms of floristics and structure, with the latter recognised more readily as a distinct group, occupying higher altitudes and dominated by *Athrotaxis cupressoides* (Jarman *et al.* 2005). Callidendrous rainforests are typically medium to tall forests dominated by *Nothofagus cunninghamii*, with *Atherosperma moschatum* present as a sub-canopy species. There is very little understorey and a very low diversity of angiosperms within these forests, however, bryophytes and lichens are plentiful and diverse (Jarman *et al.* 2005). Implicate rainforests are typically low in stature with broken, uneven canopies. The understorey is tangled and often forms a continuous layer from the ground to the canopy. Implicate rainforest tends to occupy less productive sites. Common dominant species include *Athrotaxis selaginoides* and *Nothofagus gunnii* (Jarman *et al.* 2005). Thamnic rainforest is intermediate between callidendrous and implicate rainforests (Jarman *et al.* 2005). Low-statured forests dominated by the highly fire-sensitive deciduous beech (*N. gunnii*) also frequently occur in the subalpine regions of Tasmania (Jackson 2005).

1.3.2 Mixed forest

Frequently a transition zone occurs between stands of rainforest and the adjacent non-rainforest vegetation. This ecotone is characterised by tall eucalypts above a continuous rainforest understorey (Adam 1992), and often it is of sufficient width to be recognised as a distinct

community, known as mixed forest (Bowman 2000). Mixed forests occur frequently in Tasmania, Victoria and New South Wales and are often interpreted as a rainforest invasion of tall eucalypt forest (Adam 1992). The fuel loads of mixed forests are higher than those of rainforest and the eucalypt species require fire or other canopy-removing disturbance for regeneration. If fire is absent from these communities for a sufficiently long period of time, the eucalypts will senesce and the mixed forest will succeed to rainforest (Gilbert 1959; Jackson 1968; Adam 1992; Bowman 2000). This mixture of rainforest and eucalypt forest hinders neat definitions of rainforest. In Tasmania, pure rainforest has arbitrarily been defined as having less than 5% eucalypt canopy; however, there is no evidence of any difference in stand ecology above or below this limit (Adam 1992).

1.4 Rainforest and fire in Tasmania

A mosaic of vegetation communities occurs in the rainforest rich regions of western and south-western Tasmania. Jackson (1968) proposed a model of succession, or 'ecological drift' whereby vegetation becomes increasingly flammable as the interval between fires decreases. Buttongrass moorlands, dominated by the sedge *Gymnoschoenus sphaerocephalus*, are considered the most flammable community of western Tasmania and consequently require the most frequent fire (Jackson 1968; Marsden-Smedley 1998). As communities grade from moorland into scrub and sclerophyll forest, they become increasingly less flammable and the interval between fires becomes greater (Jackson 1968). Rainforest is considered to be the climax vegetation and will frequently share boundaries with the highly flammable moorland. Despite these shared boundaries, fire in rainforest is uncommon (Jackson 1968; Barker 1991; Marsden-Smedley 1998; Read 2005). When fire does occur in rainforest, the intensity is often low (Hill 1982) and the penetration shallow. Nevertheless, on occasion, fire will sustain itself within rainforest, burning large tracts. In Tasmania this was recorded in the 1933/34 fire as well as the 1938/39 fire, where crowning was observed in rainforest (Marsden-Smedley 1998). In 1982 a large fire burned in rainforest near Savage River. In this fire, rainforest was observed burning at a variety of intensities, with peat fires, surface fires, canopy fires and scorching all recorded (Barker 1991).

The effect of fire on rainforest is complex, with the outcome depending on a number of factors, such as the intensity of the fire, the composition of the rainforest community and the vicinity of non-rainforest species to the burn (Hill 1982; Hill and Read 1984; Barker 1991). Hill and Read (1984) observed that after a low intensity humus fire in callidendrous rainforest the community

was able to regenerate to rainforest. They concluded that this was due to an absence of wet sclerophyll species within seed dispersal distance of the burned area. Conversely, in the same study, burned areas of mixed forest and rainforest adjacent to wet sclerophyll communities showed a significant reduction in rainforest species and an increase in the range and dominance of the sclerophyll component (Hill and Read 1984). This observed rise in the sclerophyll species increased the flammability of the community, with the chance of a second fire occurring before the regenerating rainforest element reached reproductive maturity becoming greater (Jackson 1968; Hill and Read 1984). Another risk of fire to rainforest is to the community composition of the forest. Although it has been shown that certain rainforest species are able to regenerate after fire, albeit at a competitive disadvantage (Mount 1979; Hill and Read 1984; Read 2005) there are other common rainforest species that show very little capacity to survive fire. These are invariably killed by a fire event (Read 2005). Many of the conifers including *Lagarostrobos franklinii*, *Phyllocladus aspleniifolius*, and *Athrotaxis selaginoides*, as well as species such as *Nothofagus gunnii*, are highly sensitive to fire. Recovery only occurs by seed dispersal from unburned stands, which can often take decades (Read 2005). Thus, a fire in rainforest containing any of these species can be disastrous (Kirkpatrick 1986). As such, it can be concluded that in Tasmania, most fires in most rainforests will have a deleterious effect.

It is clear from ecological studies (Podger *et al.* 1988), soil (di Folco and Kirkpatrick 2013) and palynological evidence (Dodson 2001) that environments now occupied by moorland and sclerophyll communities once supported rainforest. di Folco and Kirkpatrick (2013) show that since European invasion of Tasmania a dramatic shift has occurred in the distribution of rainforest and moorland in the interior of south-west Tasmania, with many sites historically occupied by rainforest now dominated by moorland. However, the authors also found that the opposite was true in the coastal regions, where rainforest is now expanding at the expense of moorland. These changes have been attributed to changing fire regimes. All of the above studies support the ecological drift theory (Jackson 1968) and provide evidence that rainforest has extensively and repeatedly burned in the past. However, the reasons why past rainforest has burned is not always known and likely to be due to many different factors, including changing climate (Dodson 2001), increased or altered human land use (Podger *et al.* 1988; di Folco and Kirkpatrick 2013) and stochastic processes. Considering the ecological importance of rainforest (Read 2005), its status as a world heritage value (Balmer *et al.* 2004) and its intrinsic worth, determining the conditions under which rainforest will burn is important.

1.4.1 Mitigating rainforest fires - hazard reduction burning

Prescribed burning is a commonly used method of reducing the chance of wildfires in areas where fire exclusion is desired. Prescribed burning is defined as the deliberate application of fire to fuels under specified conditions such that well-defined management goals are attained (Fernandes and Botelho 2003). Prescribed burning can be conducted for ecological management purposes (ecological burning), for fuel reduction (hazard-reduction burning) or for a combination of both. Ecological burning is conducted to provide the requirements for fauna or flora that require particular fire regimes (Askey-Doran 1995; Marsden-Smedley 1998). While the aims of hazard-reduction burning are primarily asset protection, ecological outcomes are often met as hazard-reduction burns aim to broaden the weather conditions under which effective fire suppression can be performed and to provide a landscape within which effective fire management can be maintained (Marsden-Smedley 1998; King *et al.* 2006). This has ecological advantages for fire sensitive species and communities, as an increase in hazard-reduction burning leads to a decrease in the amount of fire-sensitive vegetation burned (King *et al.* 2006). Both facets of prescribed burning generally aim to create a landscape mosaic of various fire ages.

Hazard-reduction burning has been shown to be an effective means of reducing the chance of landscape scale fire under most conditions (Fernandes and Botelho 2003; Gould *et al.* 2007). However, due to the requirement that hazard-reduction burns be conducted in marginal weather conditions, so as to decrease the chance of fire escape, many burns are ineffective in adequately reducing fuel (Fernandes and Botelho 2003). In Tasmania, buttongrass moorland is one of the major vegetation communities in which hazard-reduction burning is undertaken (King *et al.* 2006; Marsden-Smedley 2009). The implementation of hazard-reduction burning in buttongrass moorland aims to remove 70% of the flammable fuel from 70% of the site (Marsden-Smedley *et al.* 1999). Modelling by King *et al.* (2006) has shown that performing hazard-reduction burns in buttongrass moorlands can substantially reduce the amount of fire sensitive vegetation burned in unplanned wildfires. They propose a prescribed burning regime of 5-10% of moorland annually, to achieve the multiple management objectives of reducing fire size and fire intensity, maintaining fire intolerant vegetation and maintaining biodiversity (King *et al.* 2006). However, in order to achieve these outcomes the burning must be effective. Marsden-Smedley (1998) has modelled the prescriptions for hazard-reduction burning in buttongrass moorland and identified two methods of burning, which are differentiated on the basis of whether secure boundaries are present or not. A secure boundary can include vegetation that is too wet to burn, roads, rivers or coastline. The presence of secure boundaries greatly increases the conditions under which

hazard-reduction burns can occur and thus their effectiveness. An obvious secure boundary for moorland hazard-reduction burning is cool-temperate rainforest, due to its cool, humid microclimate and the postulated low flammability of its component vegetation (Read 2005). A clear understanding of the conditions under which rainforest will or will not burn is therefore of paramount importance in planning and implementing burning programs. This knowledge could allow for planned burning to be undertaken under a greater range of conditions, thus extending the season in which it can occur, as well as facilitating more effective fuel removal.

1.5 Fire danger indices

1.5.1 McArthur Forest Fire Danger Index

Fire Danger Ratings (FDR) are based on local weather conditions and make predictions on fire behaviour and suppression chances if a fire were to ignite. FDRs are based on fire danger indices which differ between vegetation types. In southern Australia, fire danger indices have been developed for forest and grasslands (Luke and McArthur 1978) with the McArthur Forest Fire Danger Index (FFDI) used in the forested regions of Tasmania. The FFDI was derived from approximately 400 experimental forest fires, conducted in dry sclerophyll forests with a fuel load of up to 12 t/ha (Williams *et al.* 2001) and consequently does not necessarily apply well to other Australian forest types, such as rainforest. Inputs for the FFDI are based on standard data from Bureau of Meteorology stations based in open locations and include information about the soil dryness (the Soil Dryness Index in Tasmania), temperature, humidity and wind speed data. There is no explicit calculation of fuel moisture within the FFDI, instead fuel moisture is calculated implicitly by the inclusion of air temperature and relative humidity (Matthews 2009) while the inclusion of a Drought Factor provides information about long term rainfall occurrence and drying effects (Matthews 2009).

1.5.1.1 Soil Dryness Index

The Soil Dryness Index (SDI; Mount 1972) is a predictive tool that can be used as an indicator of fuel dryness. It is currently used to assess conditions under which hazard reduction burning can take place as well as being a component of the FFDI. As the dryness of the soil affects the

dryness of the fine litter layer on its surface, a moist soil profile will result in a moist litter layer, with a dry soil profile resulting in a dry litter layer. Furthermore, soil dryness affects the rate of transpiration through vegetation (Mount 1972) with the rate of transpiration decreasing as the soil profile becomes increasingly drier. If water loss continues, green leaves may eventually die, increasing the fuel flammability (Mount 1972). Thus, the dryness of the soil profile has an important effect on fire behaviour, increasing the flammability of the surface and canopy fuels. The SDI is based on the North American Keetch-Byram Drought Index (KBDI; Keetch and Byram 1968). This index was modified to derive the SDI using Australian hydrologic research conducted largely by Bell and Gatenby (1969). The SDI is driven by the meteorological variables of rainfall and maximum temperature with the derived value corresponding to the amount of rain required (in millimetres) to return the soil profile to field capacity (Chladil and Nunez 1995). However, soil dryness can be influenced by factors other than run-off, such as soil type and soil hydrophobocity (Sullivan 2001). The KBDI was originally incorporated into the FFDI (Luke and McArthur 1978), however, in some Australian states, including Tasmania, the SDI is used instead of the KBDI. Sullivan (2001) lists a number of improvements in the SDI compared to the KBDI. These include different interception classes for different canopy densities and types of vegetation and understorey, an allowance for the drying of the canopy following rain on wet days and changes to the subsequent interception, the inclusion of flash run-off as a result of storms and showers, the incorporation of seasonal differences in evapo-transpiration rates, and the lack of assumptions about evapo-transpiration and annual rainfall.

1.5.1.2 Drought Factor

The Drought Factor (DF) is another component of the FFDI and estimates fuel moisture content from recent significant rainfall through direct wetting by rain and through wetting from below via soil moisture, which is represented by the inclusion of the SDI into the DF. Therefore, the accurate prediction of the current soil moisture is important for predicting the current fuel moisture status (Finkele and Mills undated). The DF ranges from 0 – 10 and was developed to predict the amount of fine fuel which is available to be consumed in the flaming front of a fire (McCarthy 2003). The number is intended to correspond to the percentage of fine fuels available to burn. For example, a DF of 5 predicts that 50% of the fine fuels will burn (McCarthy 2003). However, studies attempting to corroborate the predictions of the DF have found that fire intensity is as significant in influencing fuel consumption as fuel moisture content (McCarthy

2003). When the DF is calculated from the SDI higher values are produced than when the KBDI is used. This is due to a much larger evapo-transpiration term for the SDI in the water balance equation (Finkele *et al.* 2006).

In addition to the long term wetting component represented by the SDI, the DF incorporates a component based on the number of days since rain during the previous 20 day period (event age) and the amount of the last fall. The rainfall amount is defined within a set of consecutive days, each with rainfall above two millimetres (McArthur 1973; Finkele *et al.* 2006). However McArthur (1973) is not explicit about how event age should be calculated. Forestry Tasmania has determined that the event age be defined as the number of days since the day with the largest daily rainfall amount within the rain event, and this approach has been adopted by most States using the SDI to calculate DF (Finkele *et al.* 2006). However, another interpretation of event age is to simply sum the number of consecutive days since the last rain fell (Finkele *et al.* 2006).

1.5.2 Canadian Forest Fire Weather Index

The Canadian Forest Fire Weather Index (FWI) is based on the effects of weather parameters on forest floor fuel moisture conditions and generalised fire behaviour (Figure 1.2). It was developed from observations in a jack pine stand (Dowdy *et al.* 2009). The FWI is a primary input into the Canadian Forest Fire Behaviour Prediction System (FBP), which provides information on rate of spread, fuel consumption, rate of perimeter growth and head, flank and back fire spread distances (Dowdy *et al.* 2009). Inputs of the weather observations provide information on the moisture levels of three classes of forest fuel; fine fuel (FFMC), duff (DMC) and deep compact organic matter, or 'drought code' (DC; Figure 1.2). Each of the fuel moisture codes represents different fuel loads and drying times (Van Wagner 1987). These fuel moisture codes combine, along with wind speed, to provide two fire behaviour indices, the Initial Spread Index (ISI) and the Build Up Index (BUI), which themselves combine to provide the FWI (Van Wagner 1987; Figure 1.2).

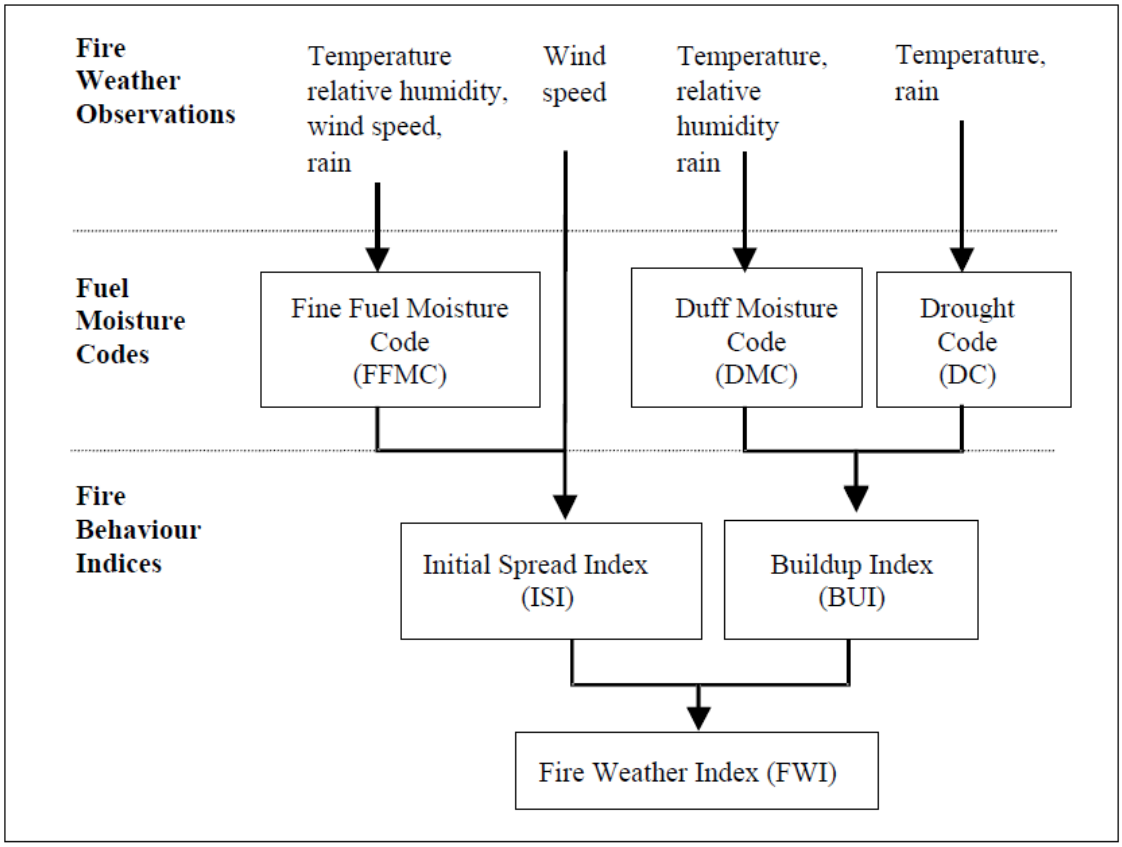

Figure 1.2 Structure of the Canadian Forest Fire Weather Index System (Van Wagner 1987).

Comparisons of the FFDI and FWI have been made by Dowdy *et al.* (2009) and Matthews (2009). Dowdy *et al.* (2009) found that the FFDI and FWI were similar to each other in that they are most sensitive to wind speed, followed by relative humidity, then temperature and least sensitive to drought. The same study found that the two indices tended to be complementary, with the FWI being more sensitive to wind speed and rainfall and the FFDI being more sensitive to relative humidity and temperature, suggesting that it may be useful for fire managers to examine both indices.

1.6 Thesis aims

The major aim of this research is to understand the climatic parameters that allow fire to ignite and sustain in rainforest.

By addressing the major research aim of this study the data collected will be able to assist in both preventing wildfires and aiding in fire suppression efforts when a wildfire becomes established.

In particular, it will provide an understanding of the conditions under which rainforest does not burn that will aid in the use of cool-temperate rainforest as a boundary for planned hazard-reduction burns, and will be used to help provide priorities for protection of natural assets during wildfires.

1.7 Structure of thesis

Chapter 2 provides a description of each of the study sites and the data.

Chapter 3 uses laboratory experiments to examine the flammability of common rainforest species and compares these to species from the adjacent sclerophyllous forest communities. Field experiments are used to validate these results.

Chapter 4 examines the spatial effects of rainforest canopy characteristics on the interception and distribution of precipitation within a rainforest.

Chapter 5 examines how the rainforest microclimate differs from that of adjacent open areas.

In the context of the previous chapters, Chapter 6 provides an analysis of the effectiveness of the SDI as a soil moisture prediction tool.

Chapter 7 describes the weather conditions associated with fires in or adjacent to rainforest and uses these data to determine conditions that have resulted in wildfire sustainability within rainforest.

Chapter 8 draws out the theoretical and practical implications of the thesis and presents ideas for further research.

Chapter 2 – Study sites

2.1 Site selection and establishment

Four sites were selected in southern and western Tasmania (Figure 2.1). The criteria for selection were: 1) to represent subalpine, callidendrous and implicate rainforest; 2) to represent both mineral and organic soils; 3) to have a nearby open space to establish a control weather station and; 4) to not be visible to the public but with quick access to the site.

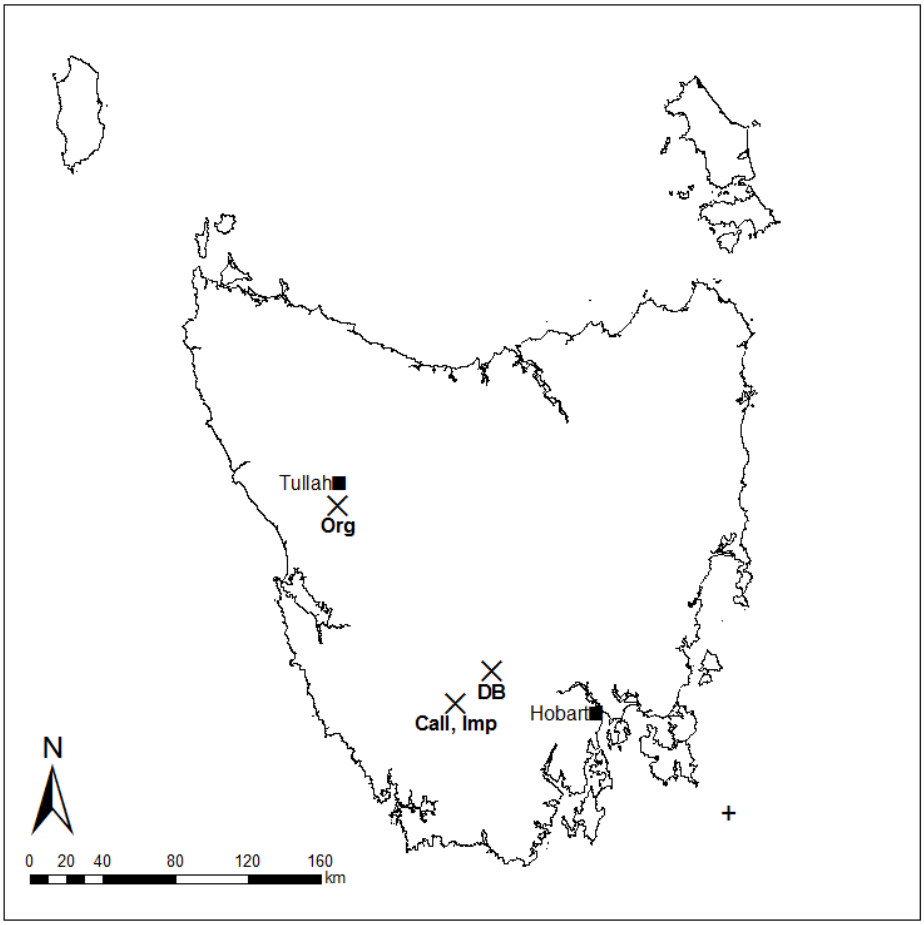

Figure 2.1 Location of study sites (crosses) within Tasmania and nearby localities (squares). + is at GDA94 55G E600000; N5196000.

Table 2.1 Site variables.

Site	Vegetation community	Location	Co-ordinates (GDA 94)	Climatic variable	Altitude (m a.s.l.)	Geology	Established
Call	Callidendrous rainforest	Creepy Crawly nature trail	449598 5257618	Temperature, relative humidity (RH), soil moisture, soil temperature, solar radiation, wind speed/direction, rainfall*	580	Mafic volcanoclasts	29/6/2010 17/1/2012 14/12/2012
Imp	Implicate rainforest	Near Creepy Crawly nature trail	449935 5281751	Temperature, RH, soil moisture, soil temperature, solar radiation	565	Mafic volcanoclasts	29/6/2010
DB	Deciduous beech montane rainforest	Lake Fenton	469261 5275392	Temperature, RH, soil moisture, soil temperature, solar radiation, wind speed	1,030	Jurassic dolerite	29/6/2010
Org	Implicate rainforest on organic soil	Mount Murchison	384353 5366789	Temperature, RH, soil moisture, soil temperature, wind speed/direction	580	Pleistocene glacial deposits	23/11/2010
Control 1	Buttongrass moorland	Mueller Road	452251 5259727	Temperature, RH, soil moisture, soil temperature, solar radiation, wind speed/direction, rainfall	490	Dolomite, diamictite and mudstone	29/6/2010 17/1/2012 14/12/2012
Control 2	Low montane heath and disturbed open ground	Lake Fenton	469284 5275058	Temperature, RH, soil moisture, soil temperature, solar radiation, wind speed/direction, rainfall	1,020	Jurassic dolerite	29/6/2010
Control 3	Buttongrass moorland	Mount Murchison	384134 5366777	Temperature, RH	580	Pleistocene glacial deposits	29/1/2011

* Rainfall data were collected from Site Call during the second and third collection periods only.

2.1.1 Site Call

Site Call consisted of callidendrous rainforest dominated by *Nothofagus cunninghamii* and *Atherosperma moschatum*. The site was at 580 m a.s.l. on Cambrian mafic volcanoclastic sandstone-siltstone-limestone of the Cleveland-Waratah Association (Mineral Resources Tasmania 1983). The canopy was about 25 m high. There was very little cover of vascular plants in the understorey; however, bryophytes and lichens were common.

An automatic weather station was established at the site on 29 June 2010 and ran until 7 April 2011. Climatic variables were logged on a Campbell Scientific CR10X data logger. Climatic variables recorded at the site included air temperature and relative humidity using a Vaisala HMP50 sensor; soil moisture using a CS616 water content reflectometer; soil temperature using a Campbell Scientific 107 temperature sensor; solar radiation using a LI-COR LI200X pyranometer; and wind speed and direction using a Met One 034B anemometer. Wind variables were measured at two metres above the ground.

The automatic weather station was re-established on 17 January 2012 and ran until 28 May 2012, then again on 14 December 2012 and ran until 8 February 2013. The climatic variables collected were as above except that they were logged on a Campbell Scientific CR1000 data logger and rainfall was collected from a Hydrological Services TB4MM tipping bucket rain gauge that can measure rainfall in 0.2 mm increments.

2.1.2 Site Imp

Site Imp consisted of implicate rainforest, with a dense and tangled understorey. The site was dominated by *N. cunninghamii* and *Anodopetalum biglandulosum*. The site was at 565 m a.s.l. on Cambrain mafic volcanoclastic sandstone-siltstone-limestone of the Cleveland-Waratah Association (Mineral Resources Tasmania 1983). The broken and uneven canopy was about 15 m tall.

An automatic weather station was established at the site on 29 June 2010 and ran until 27 December 2010. Climatic variables were logged on a Campbell Scientific CR10X data logger, with temperature and relative humidity logged by four Thermocron ibutton sensors housed in homemade Stevenson screens made from white plastic buckets with holes drilled in the side. The temperature and relative humidity data were recorded until 7 April 2011. The ibutton sensors

were established within a two metre radius of the weather station. Climatic variables recorded by the weather station were soil moisture, using a CS616 water content reflectometer; soil temperature, using a Campbell Scientific 107 sensor; and solar radiation, using a LI-COR LI200X pyranometer.

2.1.3 Site DB

Site DB was a deciduous beech (*Nothofagus gunnii*) montane rainforest. The site was at 1,030 m a.s.l. on Jurassic dolerite (Mineral Resources Tasmania 2008). The canopy was about 5 m high. There was a sparse understorey which largely consisted of *Bauera rubioides* and *Richea pandanifolia*. Due to the thick, tangled form of *N. gunnii*, there was a consistent density of vegetation from ground level to the canopy.

An automatic weather station was established at the site on 29 June 2010 and ran until 7 April 2011. Climatic variables were logged on a Campbell Scientific CR10X data logger with temperature and relative humidity logged by four Thermocron ibutton sensors, also housed in homemadeStevenson screens. These sensors were established within a two metre radius of the weather station. The climatic variables that were recorded by the weather station were soil moisture, using a CS616 water content reflectometer; soil temperature, using a Campbell Scientific 107 sensor; solar radiation using a LI-COR LI200X pyranometer; and wind speed using a Met One 034B anemometer. Wind speed was measured at two metres above the ground.

2.1.4 Site Org

Site Org consisted of implicate rainforest on organic soils. The dominant tree species were *N. cunninghamii* and *Eucryphia lucida*. The site was at 580 m a.s.l. on Pleistocene glacial and glaciogenic deposits (Mineral Resources Tasmania 2004). The canopy was approximately 20 m high and was broken and uneven. The understorey vegetation was very dense. It was dominated by *A. biglandulosum*, *Trochocarpa cunninghamii* and *Anopterus glandulosus*.

An automatic weather station was established at the site on 23 November 2010 and ran until 4 May 2011. Climatic variables were logged on a Campbell Scientific CR10X data logger. Climatic variables recorded at the site were air temperature and relative humidity using a Vaisala HMP50

sensor; soil moisture, using a CS616 water content reflectometer; soil temperature, using a Campbell Scientific 107 sensor; solar radiation, using a LI-COR LI200X pyranometer; and wind speed and direction using a Met One 034B anemometer at two metres above the ground.

2.1.5 Control 1

Control 1 was established along Mueller Road, approximately 3,000 m from Sites Call and Imp. The site was established in buttongrass moorland, with low shrubs occurring approximately 30 m to the south and taller trees occurring approximately 50 m to the north. There was no noticeable tall vegetation to the east or west of the site. The site was on Precambrian dolomite, diamictite and mudstone (Mineral Resources Tasmania 1983). The altitude of the site was 490 m a.s.l.

An automatic weather station was established on 29 June 2010 and ran until 7 April 2011. Climatic variables were logged on a Campbell Scientific CR1000 data logger. Climatic variables recorded at the site were air temperature and relative humidity, using a Vaisala HMP50 sensor; soil moisture, using a CS616 water content reflectometer; soil temperature, using a Campbell Scientific 107 sensor; solar radiation, using a LI-COR LI200X pyranometer; wind speed and direction using a Met One 034B anemometer; and rainfall, from a Hydrological Services TB4MM tipping bucket rain gauge that measures rainfall in 0.2 mm increments. Wind variables were measured at two metres above the ground.

The automatic weather station was re-established on 17 January 2012 and ran until 28 May 2012 and then again on 14 December 2012 and ran until 8 February 2013. The climatic variables collected were as above.

2.1.6 Control 2

Control 2 was established at Lake Fenton in an enclosure housing a defunct weather station that was previously operated by Hobart Water. Trees and shrubs were present within 10 m of the enclosure. The site was at 1,020 m a.s.l. on Jurassic dolerite (Mineral Resources Tasmania 2008).

The automatic weather station was established at the site on 29 June 2010 and ran until 7 April 2011. The station recorded identical measurements to Control 1, the only difference being that

air temperature and relative humidity were recorded with a Vaisala HMP45C sensor rather than an HMP50 sensor.

2.1.7 Control 3

Control 3 was established on a slope dominated by buttongrass moorland that occurred adjacent to Site Org. The geology and elevation were identical to those at Site Org.

Control 3 was established on 29 January 2011 and ran until 4 May 2011. The only measurements that were taken at Control 3 were temperature and relative humidity, which were recorded with four Thermocron ibutton sensors, housed in identical Stevenson screens as described above.

2.2 Measurements of fuel moisture

At all sites fuel moisture measurements were made between 2 February 2011 and 7 April 2011. Fuel moisture was measured with fuel moisture sticks (Forestry Tasmania 2005). Three fuel moisture sticks were established at each experimental site and one fuel moisture stick was established at each control site. Sticks were placed within a two metre radius around each weather station on wire supports so that they did not make direct contact with the ground. Fuel moisture sticks were weighed for percentage moisture on every site visit during their establishment. This was roughly every three weeks for sites Call, Imp and DB and Controls 1 and 2 and every two months for Site Org and Control 3.

The summer of 2010/2011 was both wetter and cooler than average (Bureau of Meteorology 2012*a*). As a result the rainforests remained damp and the fuel moisture sticks within the rainforest always recorded a greater weight than the scales were able to record.

2.3 Automatic weather station measurements

At each site all sensors were scanned every 10 seconds with averages, maximums or totals recorded hourly, except for wind variables, which were recorded every 15 minutes (Table 2.2). All times recorded are in eastern standard time (EST).

Table 2.2 Automatic weather station variables measured and instruments used for each site. C1 = Control 1; C2 = Control 2; C3 = Control 3.

Climatic variable	Sensor	Data type	Record (mins)	Sites
Air temperature (°C)	Vaisala HMP50	Average, Maximum	60	Call, Org & C1
Relative humidity (%)	Vaisala HMP50	Maximum	60	Call, Org & C1
Air temperature (°C)	Vaisala HMP45C	Average, Maximum	60	C2
Relative humidity (%)	Vaisala HMP45C	Maximum	60	C2
Air temperature (°C)	Thermocron ibutton	Sample	60	Imp, DB, Org & C3
Relative humidity (%)	Thermocron ibutton	Sample	60	Imp, DB, Org & C3
Volumetric soil moisture (measured at 30 cm) (%)	CS616	Average	60	Call, Imp, DB, Org, C1, C2 & C3
Soil temperature (measured at 10 cm) (°C)	Campbell Scientific 107	Average	60	Call, Imp, DB, Org, C1, C2 & C3
Solar radiation (kw/m^2)	LI-COR LI200X	Average, Maximum	60	Call, Imp, DB, C1, C2 & C3
Solar radiation (MJ/m^2)	LI-COR LI200X	Total	60	Call, Imp, DB, C1, C2 & C3
Rainfall (mm)	Hydrological Services TB4MM	Total	60	C1, C2 & C3
Wind speed (m/s)	METONE 034B	Average	15	Call, DB, Org, C1, C2 & C3
Wind direction (°)	METONE 034B	Sample	15	Call, Org, C1, C2 & C3

Each station was downloaded before the data logger memory was filled, which was every 24 days for Controls 1 and 2 and every 60 days for the remaining stations. At each download the battery was replaced with a freshly charged battery. The ibuttons were downloaded and reset roughly every 85 days.

On occasions the weather stations failed to record data between visits.

Chapter 3 – Flammability of rainforest components

3.1 Introduction

Rainforest is widely considered to be less flammable than other vegetation types (Jackson 1968; Mount 1979; Bond and van Wilgen 1996; Bowman 2000). Comparisons of the flammability of rainforest components with that of components of adjacent ecosystems have shown lower flammability in some rainforest trees in some locations (e.g. Dickinson and Kirkpatrick 1985) but no difference elsewhere (e.g. Bowman and Wilson 1988).

The flammability of species and litter is critical in predicting the conditions under which rainforest of different types will burn. Flammability consists of four components; ignitability, sustainability, combustability and consumability (Anderson 1970; Martin *et al.* 1994). These components are not independent of one another (Martin *et al.* 1994). For example, sustainability, combustibility and consumability are irrelevant in the absence of ignition (White and Zipperer 2010).

Ignitability refers to the delay until ignition occurs, and is the time to first flaming from the time of first exposure to an ignition source (Gill and Moore 1996). Sustainability relates to the ability of fuel to continue burning once ignited, independent of the initial heat source (White and Zipperer 2010). Combustibility describes how rapidly the fuel burns after ignition and is related to the rate of spread and heat of combustion (Behm *et al.* 2004). Consumability refers to the proportion of mass or volume that is consumed in the combustion process (White and Zipperer 2010). High consumability has been associated with fine fuel biomass and volatile solids (Behm *et al.* 2004).

Moisture content provides the most dominant influence on foliage flammability, with increased levels of moisture content decreasing flammability (Gill and Moore 1996; Etlinger and Beall 2004; Pellizzaro *et al.* 2007; Marino *et al.* 2010). Dimitrakopoulos and Papaioannou (2001) found that 73-94% of the variation in ignitability (time to ignition) could be explained by moisture content alone. The surface area to volume ratio of the foliage is also important (Etlinger and Beall 2004; Dibble *et al.* 2007). Gill and Moore (1996) found that surface area to volume ratio and moisture content together explained over 80% of the variance in ignitability, with increasing

moisture content increasing the ignition delay time, and increasing surface area to volume ratio decreasing it (Gill and Moore 1996).

Leaves are commonly the only plant component that is examined in laboratory flammability studies (e.g. Gill and Moore 1996; Dimitrakopoulos and Papaioannou 2001). This is because they provide uniformity and because they are often considered the most flammable part of the plant due to high surface area to volume ratios and the presence of volatile compounds (Dimitrakopoulos and Papaioannou 2001). Moisture content of the litter component is also a very important factor in the flammability of ecosystems. Fernandes *et al.* (2008) found that above 35% moisture content, the *in situ* litter of maritime pine forests was unable to sustain fire. Gillon *et al.* (1995) found similar results for a French pine forest, with fuel moisture levels above 30% unable to sustain ignition in a windless environment within a laboratory. In Australia, a moisture content of below 20% – 25% has been found necessary for fine eucalypt fuel to burn sustainably (Luke and McArthur 1978).

Some other contributors to flammability are heat content (Pellizzaro *et al.* 2007), chemical composition (Etlinger and Beall 2004), ash content (Dickinson and Kirkpatrick 1985), the arrangement of fuels in three dimensions (Fernandes *et al.* 2008), fuel bed porosity and fuel depth (Dibble *et al.* 2007). The flammability of plants is not only a function of their intrinsic properties but also of their environment, horticultural or management practices and life stage and all these factors vary among individuals of the same species (Gill and Moore 1996). Consistent results have not been found for these contributors to flammability. For example, Gill and Moore (1996) found that mineral content decreased flammability, whereas Etlinger and Beall (2004) found no such relationship. These discrepancies may be due to the large number of methods that have been used in testing the components of flammability as well as a lack of independence between the factors (Behm *et al.* 2004).

In addition to laboratory studies, fire sustainability experiments may take the form of small-scale experimental field test fires (Lin 1999; Beverly and Wotton 2007; Fernandes *et al.* 2008; Leonard 2009) or landscape-scale fires (Marsden-Smedley *et al.* 2001; Gould *et al.* 2007). Small-scale field-based test fires have the advantage of being inexpensive and capable of being conducted during fire weather conditions that may not be conducive to larger scale fires. While experimental laboratory test fires are advantageous for examining specific fuel properties by manipulating external influences such as weather variables (Beverly and Wotton 2007), unless the natural fuel arrangement is retained and a large enough area is allowed for fire to develop, laboratory results

are not necessarily transferable to fire sustainability in the field (Fernandes *et al.* 2008). Fire sustainability studies have been conducted in numerous vegetation types, including grassland (Leonard 2009), buttongrass moorland (Marsden-Smedley *et al.* 2001), shrublands (Weise *et al.* 2005) conifer forest surface fuels (Beverly and Wotton 2007; Fernandes *et al.* 2008) as well as tropical rainforest surface fuels (Uhl *et al.* 1988), however the capacity for fire to be sustained within cool-temperate rainforest has not been examined.

This chapter aims to determine if leaf and litter material from species that dominate cool-temperate rainforest and mixed forest are less flammable than leaf and litter material from species that occur in surrounding fire dependent ecosystems, such as *Banksia marginata* scrubland and *Eucalyptus coccifera* woodland, as well as determining if it is possible to predict flammability from other fuel properties, such as surface area to volume ratio. The chapter also looks at the fuel moisture conditions in a small-scale field based ignition experiment to see how rainforest litter responds to ignition *in situ*.

3.2 Methods

3.2.1 Study sites, site establishment and sampling procedure

The sites sampled for the laboratory component of this study are described in Chapter 2. Fresh leaf samples were mostly collected from Site Call, with additional collection taking place at Site Imp and Site DB when species of interest were absent from Site Call (Table 3.1). Species of interest for flammability studies were identified by dominance within each of the three rainforest communities and adjacent communities.

Table 3.1 Species examined for the flammability experiments and their collection location.

Species	Site
Atherosperma moschatum	Call
Eucryphia lucida	Call
Nothofagus cunninghamii	Call
Orites diversifolius	Call
Anodopetalum biglandulosum	Imp
Nothofagus gunnii	DB
Bauera rubioides	DB
Eucalyptus nitida	Scotts Peak Road
Banksia marginata	Scotts Peak Road

Fresh leaf material of each species was collected from at least four different individual plants within each community. The samples were immediately sealed in snap lock bags then placed in plastic buckets with sealing lids to minimise moisture loss during transport.

Surface litter samples were taken from three different sites. These were the callidendrous rainforest at Site Call (referred to as callidendrous forest), the *Nothofagus gunnii* dominated Site DB (referred to as deciduous beech scrub), and the *Eucalyptus coccifera* dominated woodland surrounding Site DB (referred to as *Eucalyptus coccifera* woodland). Litter samples were collected from at least 6 different locations within each vegetation type and immediately placed in plastic bags for transport back to the laboratory.

The site used for the field component of this study occurred in thamnic rainforest and was located about one hour south of Hobart, along Arve Road, west of Geeveston. The site was selected as it met the following criteria: 1) was accessible to a road for fire suppression purposes; 2) was not steeply sloped; 3) has no eucalypt litter; and 4) was within a one hour drive of Hobart.

Test fire plots were identified in the field in November 2012 and marked. Some modification of the vegetation was required in order to establish tarpaulins over the plots with the purpose of manipulating the fuel dryness. This occurred on one day in November 2012 and involved pruning of saplings and other scrub but did not affect the litter layer. Most plots in which test fires were to be carried out were covered by a 4.9 by 6.1 metre tarpaulin, which was pitched roughly one metre above the ground. Tarpaulins were established throughout November and December 2012 and January and February 2013. This resulted in 9 plots with different durations since it last rained.

In order to estimate the fuel load of each plot without destroying it, a double sampling approach was used (Catchpole and Wheeler 1992; Catchpole and Catchpole 1993). Cover and height of leaf litter, branch litter less than 6 mm in diameter, branch litter greater than 6 mm in diameter and bare ground/roots were recorded for 10 one by one metre quadrats. The fuel within these quadrats was then collected and taken to the laboratory where it was dried at 105 °C for 48 hours. The following equation was then used to determine percent moisture content (van Wilgen *et al.* 1990; Gill and Moore 1996; Behm *et al.* 2004):

Percent moisture content = (*mwet* – *mdry*)/*mdry* * 100

Equation 3.1

where *mwet* is the mass of the litter batch before drying, and *mdry* is the mass of the litter batch after drying. Each sample was then sorted into leaf litter, branch litter less than 6 mm in diameter and branch litter greater than 6 mm in diameter, with each component weighed separately.

3.2.2 Laboratory flammability experiments

Flammability experiments were undertaken on the fresh leaf material collected from the live plants, and surface litter, which consisted of woody branch and bark material. All leaf material was excluded from the litter samples as leaves of varying dryness were examined separately.

3.2.2.1 Laboratory procedures

The samples of fresh leaf material were processed as soon as possible on return to the laboratory, which was generally about two hours after collection. For each species, 10 leaf samples were randomly selected from the original pool of material. From the same pool of material a pair was then selected for each of the original 10 samples which was of near identical size and shape. The duplicate leaves were weighed as a batch and placed in the drying oven at 105 °C for 48 hours, while the original leaves were used in the flammability experiments. This was done in order to determine an estimate of foliar moisture content from samples that were to be destroyed in the flammability experiments. The dried batch was then re-weighed to determine an estimate of percent foliar moisture content using equation 3.1.

For each single leaf sample the weight, length, area and thickness were measured. However, not all species had leaves of sufficient size to be studied individually. In cases in which the leaf was judged too small to provide sufficient data, a branchlet of leaves was examined. This was the case for *Nothofagus gunnii, N. cunninghamii* and *Bauera rubioides*. Length of all leaf and branchlet samples was measured to the nearest millimetre with a scale ruler. For *N. gunnii, N. cunninghamii* and *B. rubioides* length was considered to be from the apex of the highest leaf on the branchlet to the base of the lowest leaf. Weight was measured to the nearest gram on an electronic scale. Leaf thickness was measured to the nearest millimetre with callipers and standardised by taking a

thickness two-thirds of the way to the midrib, near the midpoint of the lamina (Gill and Moore 1996). For *N. gunnii* and *N. cunninghamii* the leaf thickness was measured as the thickness of the leading leaf two-thirds of the way to the midrib. For *B. rubioides*, thickness was recorded as the thickness of the stem, half way along the length of the branchlet. Area of each leaf was determined by scanning the samples, calculating the number of pixels and converting this number to millimetres squared. For *N. gunnii* and *N. cunninghamii* branchlets, the area was calculated individually for each sample. For all other leaf material the average area was calculated from between 18 and 77 randomly selected samples, depending on the size of the leaves, with a greater number of samples taken from smaller leaves. Area was not calculated for *B. rubioides*. The volume of each leaf was determined for *Anodopetalum biglandulosum*, *Atherosperma moschatum*, *Banksia marginata*, *Eucryphia lucida*, *Eucalyptus nitida* and *Orites diversifolius* using the following equation:

$$\text{Volume} = (SA * 2) * t$$

Equation 3.2

where SA is the mean surface area for each species and t is the thickness of the leaf sample, measured two-thirds of the way to the midrib.

Branch litter was divided into the following diameter size classes: 0-3 mm, 3.1-6 mm, and 6.1-25 mm, modified from Brown *et al.* (1982). Fresh weight, diameter and length were recorded as for the leaf samples. Volume was determined for each branch by assuming a cylindrical shape and applying the formula for a cylinder, with diameter measured from the mid-length of each sample. In addition, bark litter was prominent in the litter at the *Eucalyptus coccifera* woodland and this was used as an additional sample, with weight and length of individual pieces being recorded.

3.2.2.2 Flammability

Ten samples of each species (leaf material) or class (litter material) were measured by exposing them to an ignition source, in the form of a gas flame of constant height burning vertically within the still air of a fume cabinet (Dickinson and Kirkpatrick 1985). Leaf material was held in a natural position with tongs, with the tip of the vegetation being subjected to the flame. Woody material was held at a slight angle to the horizontal, with the end of the branch exposed to the flame. For all samples, the rate of spread and time to ignition were recorded with a stopwatch.

Time to ignition was defined as the amount of time required until flames appeared on the leaves. Flaming was not necessarily sustained. Where ignition was not sustained, the time to full combustion whilst being subjected to the flame was recorded ("time to consumption"). The nature of burning (sparking, spitting, smoking, smouldering) was noted.

3.2.2.3 Moisture content

The effect of moisture content on the flammability of the fresh leaf samples was tested by repeating the above procedures on samples of varying moisture content. Samples were dried slowly by being placed in a plant press with samples then subjected to the flammability tests described above every 18, 24, 42, 48, 72 and 120 hours. At each sampling time, paired samples were taken to determine the percent foliar moisture content using the oven drying method described above and equation 3.1.

The woody litter samples were air dried on a drying rack in the laboratory, where the daily air temperature was an average of 23.5 °C and the mean relative humidity was 44%. Ten samples of each diameter size class were tested periodically for time to sustained ignition, which was recorded with a stopwatch. The bark sample was only tested once as 100% of the samples ignited. After each test the weight, length and diameter were measured for each sample as well as the combined weight of all samples in each size class. This was used to determine the moisture content for each size class sample after oven drying for 48 hours at 105 °C and using Equation 3.1. Once woody litter samples had been oven-dried to determine moisture content, 10 samples of each size class were tested for rate of spread, using the same methods as for leaves. This provided the rate of spread at zero percent moisture content.

The woody litter samples generally consisted of the branch material of the dominant tree in each habitat; however there was also some branch litter from two additional identifiable species. These were *A. moschatum*, in the callidendrous forest, and *B. rubioides*, from the deciduous beech scrub. Notes were made when samples came from one of these two species.

3.2.3 Field flammability experiments

3.2.3.1 Test fire procedure

Before any fires were ignited, fuel data were collected from the plot with litter height and cover recorded. Fuel continuity was assessed by assigning a score between one and three, with one being low fuel continuity and three being fuel of continuous cover and height. Fuel moisture was measured by collecting litter samples within each one by one metre burn plot, prior to ignition. These samples were sealed in tins and transported back to the laboratory as soon as the day's work had concluded. The fuel samples were then weighed before being dried at 105 °C for 48 hours. Fuel moisture was then determined using equation 3.1. Soil moisture was assessed in an identical way from on average three samples under each tarpaulin, collected just below the litter layer. The soil moisture content of each of the three samples was averaged to give one value for all the plots under each tarpaulin.

Test fires were conducted in a one by one metre plot within the area covered by the tarpaulin. A total of 19 sustainability test fires were conducted across all plots on 26 March 2013 and carried out by the Parks and Wildlife Service and Forestry Tasmania in conditions which were unsuitable for the non-manipulated forest to carry fire. This ensured that no undesired fires occurred during the experiment. A drip torch containing a 2:1 diesel/petrol mixture was used to create a one metre ignition line along one edge of the test fire plot. Ignition lines were located along the down slope edge of the plot to provide the fire with the best chance of sustaining. A fire was judged to have sustained if it remained alight for at least 5 minutes or travelled one metre to the opposite end of the plot.

3.2.3.2 Weather data collection

Wet and dry bulb temperature was measured using a whirling psychrometer immediately prior to each test fire. Relative humidity was then calculated from these data. Wind speed was measured with a hand held anemometer at approximately 1.5 m above ground level, and the average wind speed for the duration of the test fire was recorded. As it was thought that the wind speed would be almost zero within the forest, the Beaufort scale was used to record the wind speed in the canopy (Table 3.2). This was done prior to each test fire.

Table 3.2 Beaufort scale for estimating wind velocity (Luke and McArthur 1978).

Beaufort number	Term	Description	Wind speed (km/h)
0	Calm	Smoke rises vertically.	<1
1	Light air	Smoke drifts slowly; wind vanes not affected.	1-5
2	Light breeze	Wind felt on face; leaves rustle; ordinary wind vanes move.	6-11
3	Gentle breeze	Leaves and twigs in motion; wind extends light flag.	12-19
4	Moderate breeze	Dust and loose paper raised; small branches move.	20-28
5	Fresh breeze	Small trees sway.	29-38
6	Strong breeze	Large branches in motion; whistling heard in telephone wires.	39-49
7	Near gale	Whole trees in motion; inconvenience felt when walking against wind.	50-61
8	Gale	Twigs broken off trees; progress of walkers impeded.	62-74
9	Strong gale	Branches broken off trees.	75-88

3.2.4 Data analysis

3.2.4.1 Laboratory flammability experiments

It was not possible to perform all analyses on all the leaf species as there were differences in the sampling procedures between the species when branchlets were used as the sample unit rather than individual leaves. To distinguish between these sub-groups the species in which whole leaves were sampled individually are referred to as foliar samples.

Four of the main vegetation communities of western Tasmania were given a score between one and four, with one being the most fire-prone and four being the least fire prone (Jackson 1968). Each species was then marked as being either capable or incapable of occupying a niche in that community. For each community occupied by a species, the species would be assigned that community's score, with the final species score being the sum of all the community scores divided by the number of communities in which the species occurs (Table 3.3).

Table 3.3 Community scores for all leaf species examined.

Species	Moorland (1)	Scrub (2)	Mixed forest (3)	Rainforest (4)	Aggregate score	Species score
Anodopetalum biglandulosum	-	X	X	X	9	3.0
Atherosperma moschatum	-	-	X	X	7	3.5
Banksia marginata	X	X	X	-	6	2.0
Bauera rubioides	X	X	X	X	10	2.5
Eucryphia lucida	-	-	X	X	7	3.5
Eucalyptus nitida	X	X	X	-	6	2.0
Nothofagus cunninghamii	-	-	X	X	7	3.5
Nothofagus gunnii	-	-	-	X	4	4.0
Orites diversifolius	-	X	X	X	9	3.0

During the time to ignition experiments on the leaf species, some samples failed to ignite and an arbitrary value of 25 seconds was assigned to these. Twenty five seconds was chosen as this value was slightly longer than the longest time to ignition for any species (19.07 seconds).

The residuals of a linear regression between rate of spread and moisture content for all leaf samples were plotted against each leaf species. One-way analysis of variance (ANOVA) with Tukey's multiple range comparison test was then performed to determine significant differences between species. Spearman rank correlation was performed on the residual from the relationship between rate of spread and moisture content and the community score.

General linear modelling using the default options in Minitab 16 was used to determine variables that significantly contributed to rate of spread of the foliar leaf samples. All theoretically reasonable combinations of predictor variables were tested. The best model was selected on the basis of the adjusted *R*-sq. A similar test was conducted for the time to ignition of the foliar leaf species.

Leaf species moisture content at each sampling time was expressed as a percentage of the original foliar moisture content. Loss was then determined by subtracting this value from 100. The amount of moisture loss as a percentage of the initial moisture content was regressed on the rate of spread of the flame front.

One-way ANOVA with Tukey's multiple range comparison test was used to examine the relationship between surface area to volume ratio (SA/V) of the foliar samples between different species. Shapiro-Wilks test was used to test whether the surface area to volume ratio (SA/V) data were normal and a Fligner-Killeen test was used to test for homogeneity of variances. The data were found to be not normal and the variances were unequal, therefore a Kruskal-Wallis rank sum test was used to test for differences in SA/V between species, followed by pairwise Wilcoxon rank sum tests with Bonferroni adjustments.

Pearson's product moment correlation coefficient was used to test correlations between SA/V and rate of spread for foliar leaf species individually over all moisture contents as well as for foliar leaf species as a group at the 72 hour, 120 hour and oven-dried drying periods. Below the 72 hour drying period not all the species were sufficiently dried to record a rate of spread.

For each leaf species regression analysis was used to examine the relationship between time to consumption and percentage moisture content. The *N. gunnii* sample was influenced by an outlier. A test was performed that omitted this outlier.

Dead fuel moisture content (DFMC) of the litter samples was plotted against hours drying to determine the nature of the relationship between DFMC and the rates of drying between samples. Individual scatterplots were used to visualise the relationship between time to ignition and the hours drying. Hours drying, rather than percentage moisture content, was chosen as the variable as it enabled the x-axes of the graphs to remain constant between species, thus providing greater ability to visualise the differences in the time to ignition.

One-way ANOVA with Tukey's multiple range comparison test was used to test any differences between rate of spread of the litter from the three different vegetation types, rate of spread of the fine fuels (0-6 mm size classes as well as bark) from the three different vegetation types, and differences in the size classes from within vegetation types. The relationship between rate of spread and SA/V of the litter was examined using linear regression.

The Kruskal-Wallis *H*-test was used to determine if the time to sustained ignition for all fuel moisture contents in the fine fuel classes differed between vegetation types. The Kruskal-Wallis test was used in preference to one-way ANOVA as the distribution of the residuals was non-normal in ANOVA. Differences in the time to sustained ignition of litter for each size class within the vegetation types was tested using one-way ANOVA with Tukey's multiple range comparison test then plotted on interval plots.

3.2.4.2 Field flammability experiments

Linear regression was used to derive a relationship between fuel load and the best predictor of fuel load. Because of difficulties in implementation, there were insufficient data for statistical analysis of the test fires, of which only four managed to sustain fire.

3.3 Results

3.3.1 Leaf samples

There was no relationship between the derived community score and the residuals from the regression of rate of spread with moisture content (Spearman rank correlation coefficient = 0.062; $P = 0.865$).

Bauera rubioides and *N. gunnii* had a rate of spread that was much slower than would be expected for their respective moisture contents (Figure 3.1). *Eucalyptus nitida*, on the other hand, recorded the fastest rate of spread for its moisture content (Figure 3.1). Interestingly, the non-rainforest species, *Banksia marginata,* did not display a particularly fast rate of spread at any moisture content, as would have been expected (Figure 3.1). *Banksia marginata* displayed no significant difference from many of the rainforest and mixed forest specialists, such as *A. moschatum* and *N. cunninghamii* (Figure 3.1, Table 3.4).

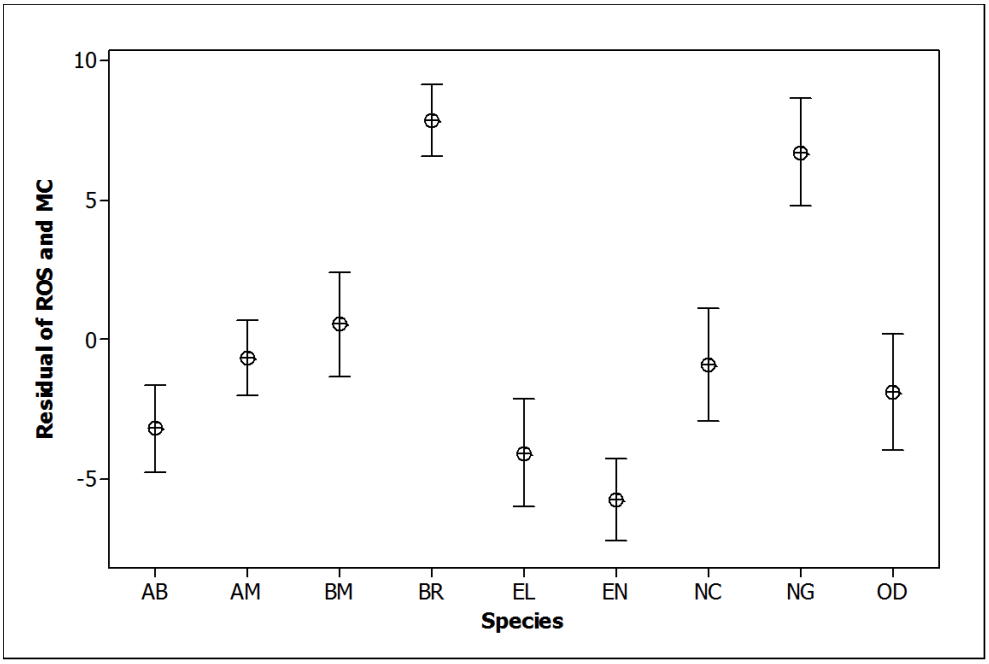

Figure 3.1 Interval plot of the residuals from the regression of rate of spread (ROS; mm s^{-1}) on moisture content (MC) for each species (95% confidence intervals). AB = *Anodopetalum biglandulosum*; AM = *Atherosperma moschatum*; BM = *Banksia marginata*; BR = *Bauera rubioides*; EL = *Eucryphia lucida*; EN = *Eucalyptus nitida*; NC = *Nothofagus cunninghamii*; NG = *Nothofagus gunnii*; OD = *Orites diversifolius*.

Table 3.4 Tukey's multiple range comparison test for species against residuals of rate of spread and moisture content. Means that share a letter are not statistically different at $P < 0.05$.

Species	Mean
Bauera rubioides	7.862 A
Nothofagus gunnii	6.734 A
Banksia marginata	0.553 BC
Atherosperma moschatum	-0.633 BCD
Nothofagus cunninghamii	-0.872 BCD
Orites diversifolius	-1.883 BCDE
Anodopetalum biglandulosum	-3.186 CDE
Eucryphia lucida	-4.055 DE
Eucalyptus nitida	-5.746 E

Moisture content, species and the interactive effect of moisture content and species were the components of the best model to predict rate of spread (Table 3.5). Arcsine transformation (Arcsine (square root (rate of spread/100))) failed to normalise rate of spread. This transformation is often used on percentage data where it is either zero inflated or 100% inflated.

Table 3.5 General linear model (log-link function) for response variables to rate of spread (mm s^{-1}) for foliar leaf species (excludes *Bauera rubioides*, *Nothofagus cunninghamii* and *N. gunnii*).

Variable	*F*-value	*P*-value
Moisture content	340.73	<0.001
Species	8.87	<0.001
Moisture content * species	11.81	<0.001

S = 7.35174 *R*-sq = 56.8% *R*-sq (adj) = 55.7% *n* = 480

Atherosperma moschatum leaves had the highest fresh picked moisture content with fresh samples averaging 211% of their oven-dry weight. The leaves of the control species *E. nitida* and *B. marginata* recorded the lowest fresh picked moisture content with average levels being 79 and 99% respectively. Of the rainforest species *E. lucida* recorded the lowest fresh picked leaf moisture at 104%.

Eucalyptus nitida leaves dried out the most rapidly following collection, whereas *B. rubioides* branchlets dried out the least rapidly. *Bauera rubioides* branchlets required the greatest amount of moisture loss before flaming was sustained (Figure 3.2). *Eucalyptus nitida* leaves required the next greatest but this was likely to have been an artefact of sampling interval and *E. nitida* is not included in Figure 3.2 and Table 3.6. *Bauera rubioides* branchlets were able to sustain flame after 48 hours of drying, when the moisture content was 35% of the oven-dry weight (Figure 3.2). This appeared to be an exception as at lower recorded moisture contents flame sustainability did not occur. Only when the samples were at 0% moisture content, determined by oven-dried weight, was flame sustainability again achieved. The leaves of *Eucryphia lucida* and *A. biglandulosum* both recorded flaming after losing 36% of their field moisture content, which was the smallest loss of moisture at flaming for any of the species sampled (Figure 3.2).

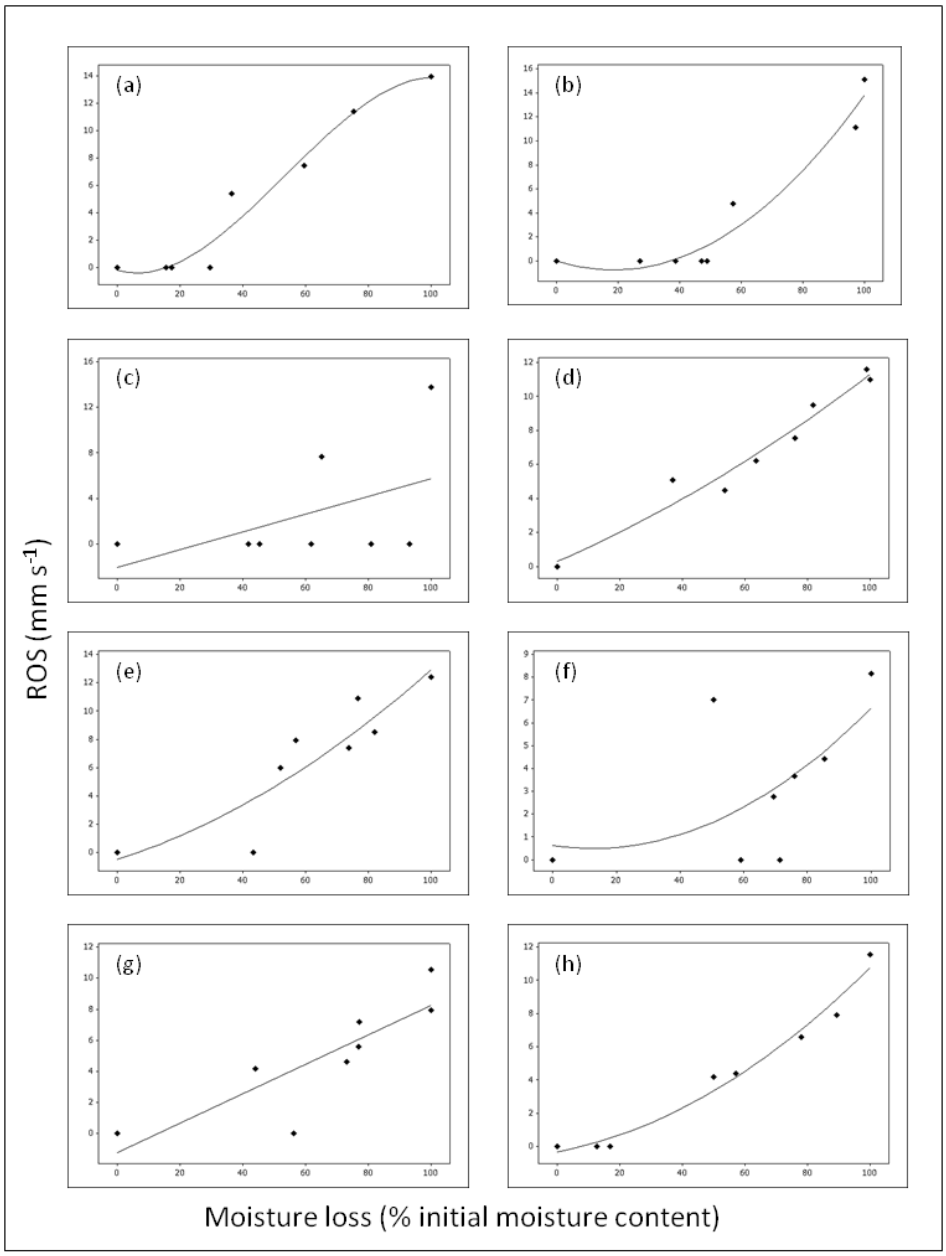

Figure 3.2 Relationship between rate of spread (ROS; mm s^{-1}) and air dry moisture loss: (a) *Anodopetalum biglandulosum*; (b) *Atherosperma moschatum*; (c) *Bauera rubioides*; (d) *Eucryphia lucida*; (e) *Nothofagus cunninghamii*; (f) *Nothofagus gunnii*; (g) *Orites diversifolius*; (h) *Banksia marginata*.

All relationships between rate of spread and leaf moisture loss were significant except for those for *B. rubioides* and *N. gunnii* (Table 3.6). The fastest rate of spread recorded for oven-dried material was for *A. moschatum* (15.15 mm s^{-1}), whereas the slowest rate of spread recorded was for *N. gunnii* branchlets (8.16 mm s^{-1}). Of the foliar samples the slowest rate of spread recorded was for *O. diversifolius* (10.55 mm s^{-1}).

Table 3.6 Regression equations, *P*-values and *R*-sq for the relationship between rate of spread (mm s^{-1}) (ROS) and percentage moisture loss (ML) of leaf species.

Species	Equation	*P*-value	*R*-sq
Anodopetalum biglandulosum	ROS = 13.91 – 0.00511 ML – 0.001404 ML^2 + 0.000006 ML^3	0.003	0.96
Atherosperma moschatum	ROS = 13.84 – 0.1698 * ML + 0.000495 ML^2	0.001	0.95
Eucryphia lucida	ROS = 11.31 – 0.1343 ML + 0.000275 ML^2	0.000	0.96
Nothofagus cunninghamii	ROS = 12.94 – 0.1780 ML + 0.000512 ML^2	0.016	0.81
Orites diversifolius	ROS = 8.267 – 0.07064 ML	0.009	0.71
Banksia marginata	ROS = 11.28 – 0.3016 * ML + 0.003231 ML^2	0.014	0.99

The surface area to volume ratio (SA/V) differed among foliar species (Figure 3.3, Table 3.7). *Anodopetalum biglandulosum* and *E. lucida* recorded the highest SA/V, while the non-rainforest species *E. nitida* recorded the lowest SA/V (Figure 3.3, Table 3.7).

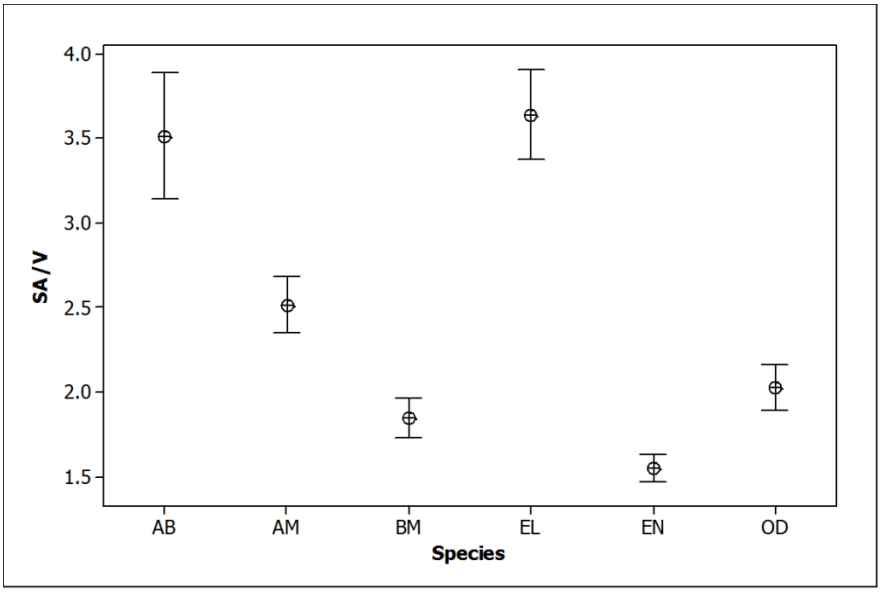

Figure 3.3 Interval plot for surface area to volume ratio (SA/V) of foliar leaf species (95% confidence intervals). AB = *Anodopetalum biglandulosum*; AM = *Atherosperma moschatum*; BM = *Banksia marginata*; EL = *Eucryphia lucida*; EN = *Eucalyptus nitida*; OD = *Orites diversifolius*.

Table 3.7 Surface area to volume ratios (SA/V) of foliar leaf species (n = 80). Medians that share a letter are not statistically different at P < 0.05 (pairwise Wilcoxon rank sum test with Bonferroni adjustment). IQR = interquartile range.

Species	SA/V	
	Median	IQR
Anodopetalum biglandulosum	2.94 A	1.78
Eucryphia lucida	3.45 A	1.39
Atherosperma moschatum	2.38 B	0.96
Orites diversifolius	1.92 C	0.51
Banksia marginata	1.79 C	0.81
Eucalyptus nitida	1.52 D	0.42

Only after 72 hours of air drying was there a significant relationship between SA/V and rate of spread when all the species were aggregated (Pearson's product moment correlation coefficient = 0.344; P = 0.022). However, when the drying times were aggregated and the species examined individually there was a significant correlation between SA/V and rate of spread for two of the foliar species, *E. lucida* and *E. nitida* (Table 3.8).

Table 3.8 Pearson's product moment correlation coefficients for surface area to volume ratio (SA/V) and rate of spread (mm s^{-1}) for the six foliar species across all drying times (n = 80). Significant correlations are shown in bold (* P <0.05; ** P <0.01; *** P <0.001).

Species	Pearson's value
Anodopetalum biglandulosum	0.495
Atherosperma moschatum	0.188
Eucryphia lucida	**0.641*** **
Orites diversifolius	0.224
Banksia marginata	-0.018
Eucalyptus nitida	**0.371** **

The time to consumption following ignition tended to increase as the moisture content of the leaves increased (Figure 3.4). *Atherosperma moschatum*, *B. rubioides*, *N. gunnii* and *O. diversifolius* showed the most resistance to burning (Figure 3.4). The relationship between time to consumption and moisture content was significant for all species except *A. biglandulosum* (Table 3.9). However, when the outlier was removed from the *N. gunnii* sample it also failed to produce a significant relationship.

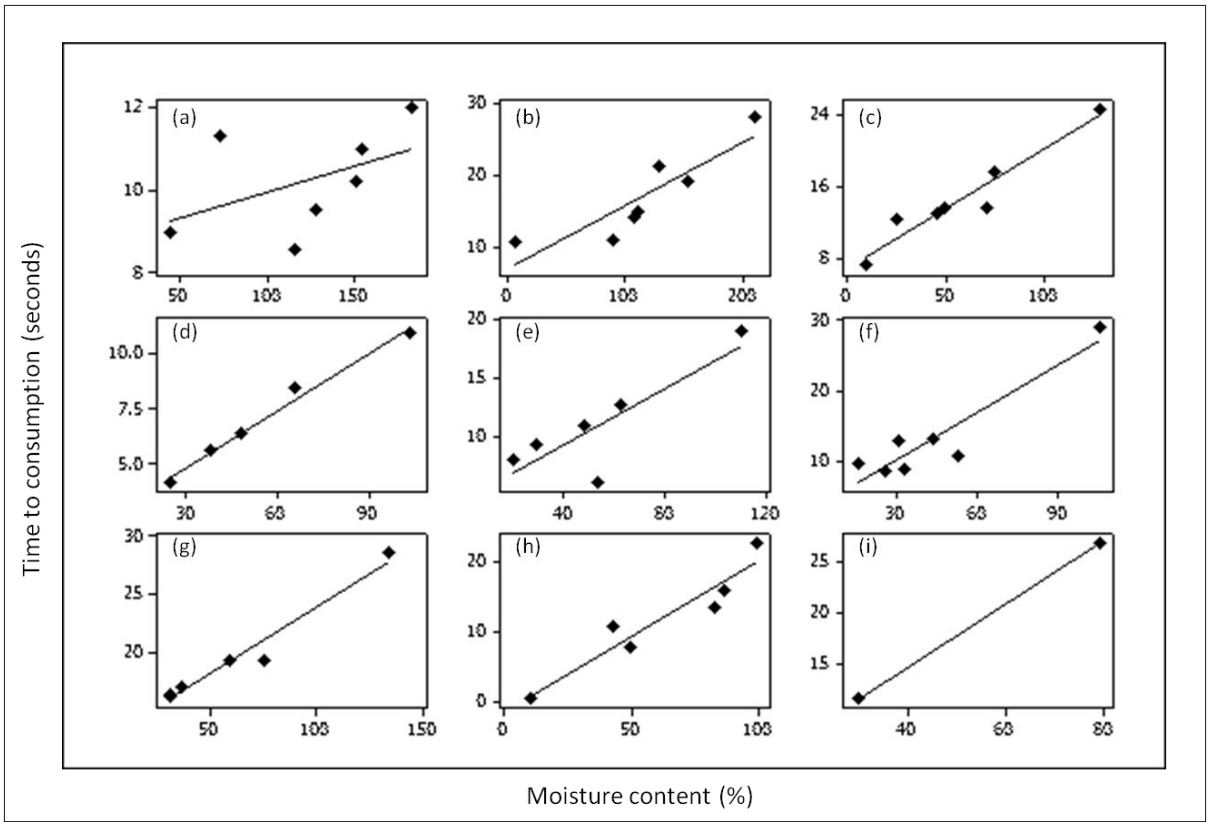

Figure 3.4 Relationship between time to consumption and moisture content of leaves of selected species: (a) *Anodopetalum biglandulosum*; (b) *Atherosperma moschatum*; (c) *Bauera rubioides*; (d) *Eucryphia lucida*; (e) *Nothofagus cunninghamii*; (f) *Nothofagus gunnii*; (g) *Orites diversifolius*; (h) *Banksia marginata*; (i) *Eucalyptus nitida*.

Table 3.9 Regression equations, *P*-values and *R*-sq for the relationship between time to consumption (TTC) and moisture content (MC) of leaf species.

Species	Equation	*P*- value	*R*-sq
Anodopetalum biglandulosum (*n* = 7)	TTC = 8.628 + 0.01291MC	0.266	0.24
Atherosperma moschatum (*n* = 7)	TTC = 6.749 + 0.08798MC	0.009	0.78
Bauera rubioides (*n* = 7)	TTC = 6.814 + 0.1340MC	0.001	0.93
Eucryphia lucida (*n* = 5)	TTC = 2.298 + 0.08609MC	0.001	0.98
Nothofagus cunninghamii (*n* = 6)	TTC = 4.481 + 0.1210MC	0.034	0.72
Nothofagus gunnii (*n* = 7)	TTC = 3.381 + 0.2271MC	0.002	0.87
Orites diversifolius (*n* = 6)	TTC = 12.40 + 0.1147MC	0.001	0.96
Banksia marginata (*n* = 6)	TTC = - 1.470 + 0.2159MC	0.004	0.90

Both foliar moisture content and species were significant in predicting time to ignition, explaining 62.19% of the variation (Table 3.10).The time to ignition varied greatly among the species. *Eucryphia lucida*, *N. cunninghamii* and *E. nitida* ignited in less than four seconds for fresh picked samples, whereas *N. gunnii* and *B. marginata* took in excess of 16 seconds before ignition was achieved. Ignition of *B. rubioides* was never achieved for the fresh picked samples (Figure 3.5). Once drying commenced the time to ignition of the different species was varied, with *N. cunninghamii* and *E. nitida* showing the fastest drying rates, with time to ignition being almost instantaneous after 42 hours of air drying (Figure 3.5).

Table 3.10 General linear model (log-link function) for response variables to time to ignition for all foliar leaf species (excludes *B. rubioides*, *N. cunninghamii* and *N. gunnii*).

Variable	*F*-value	*P*-value
Moisture content	71.66	<0.001
Species	8.19	<0.001

S = 3.14999 *R*-sq = 67.1% *R*-sq(adj) = 62.2% *n* = 71

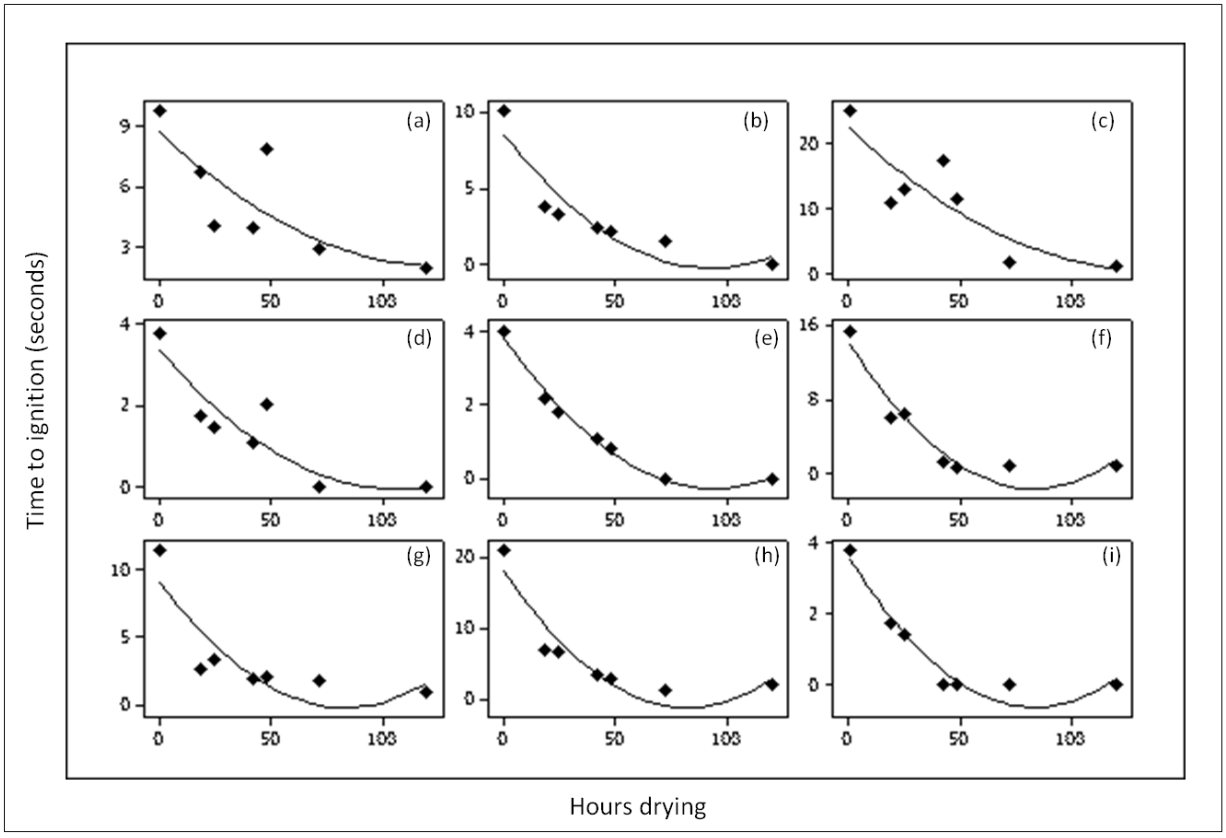

Figure 3.5 Relationship between time to ignition of leaves of selected species and hours air drying: (a) *Anodopetalum biglandulosum*; (b) *Atherosperma moschatum*; (c) *Bauera rubioides*; (d) *Eucryphia lucida*; (e) *Nothofagus cunninghamii*; (f) *Nothofagus gunnii*; (g) *Orites diversifolius*; (h) *Banksia marginata;* (i) *Eucalyptus nitida*.

3.3.2 Woody litter samples

The rate of spread for all woody litter size classes differed among the three vegetation types sampled, with the highest mean rate of spread recorded for the *Eucalyptus coccifera* woodland and the lowest for the callidendrous forest (Table 3.11). The callidendrous forest and *Eucalyptus coccifera* woodland litter were significantly different from one another, with the deciduous beech scrub being statistically identical to both the other types (Table 3.11). There was no significant difference in the mean rate of spread in the fine fuels (0-6 mm) among vegetation types (Table 3.11).

Table 3.11 Mean values and results of one-way ANOVA for litter size classes and rate of spread (mm s^{-1}) between different vegetation types. The *Eucalyptus coccifera* group includes bark litter. Means that share a letter in rows are not statistically different at P <0.05.

Size class	Callidendrous	Deciduous beech	*Eucalyptus coccifera*	*F*-value	*P*-value
All sizes	1.747 A (n = 30)	1.990 AB (n = 30)	2.739 B (n = 40)	3.70	0.028
Fine fuels (0-6 mm)	2.229 (n = 20)	2.449 (n = 20)	3.119 (n = 30)	2.17	0.123

There were significant differences in the rate of spread among the three litter size classes for each vegetation type, and the bark sample in the case of the *Eucalyptus coccifera* woodland (Table 3.12). The rate of spread for the 0-3 mm litter size class was always significantly faster than the rate of spread from the 3-6 mm and 6-25 mm litter size classes. There was no significant difference between the rate of spread for and 3-6 mm and 6-25 mm size classes within each respective vegetation type (Table 3.12). For the *Eucalyptus coccifera* woodland, the bark sample was significantly different from all other litter classes, recording the fastest mean rate of spread (Table 3.12).

Table 3.12 Mean values and results of one-way ANOVA for rate of spread (mm s^{-1}) for litter fuel components in each vegetation type. Means that share a letter in rows are not statistically different at P <0.05.

Habitat	0-3mm n = 10	3-6 mm n = 10	6-25 mm n = 10	Bark n = 10	*F*- value	*P*-value
Callidendrous	2.920 A	1.537 B	0.785 B	NA	11.07	<0.001
Deciduous beech	3.251 A	1.646 B	1.073 B	NA	16.86	<0.001
Eucalyptus coccifera	3.411 B	1.631 C	1.654 C	5.778 A	16.81	<0.001

The relationship between rate of spread of litter samples and SA/V was significant for all vegetation types, with litter samples with higher surface area to volume ratios having a faster rate of spread (Table 3.13).

Table 3.13 Regression equations, *P*-values and *R*-sq for the relationship between rate of spread (ROS; mm s^{-1}) and surface area to volume ratio (SA/V) of litter samples in all size classes.

Habitat	Equation	*P*- value	*R*-sq
Callidendrous (*n* = 30)	ROS = 0.7483 + 2.331 SA/V	<0.001	0.64
Deciduous beech (*n* = 30)	ROS = 0.8015 + 3.068 SA/V	<0.001	0.49
Eucalyptus coccifera (*n* = 30)	ROS = 0.7993 + 3.434 SA/V	<0.001	0.51

For the litter samples, time to sustained ignition was recorded for each size class within each vegetation type. There was not a strong relationship between time to sustained ignition and dead fuel moisture content (DFMC) for the litter samples (Figure 3.6). The 6-25 mm diameter class in particular showed no time response, with the 0-3 mm and 3-6 mm diameter classes displaying more of a tendency towards increased time to sustained ignition with increased dead fuel moisture content (Figure 3.6; Table 3.14).

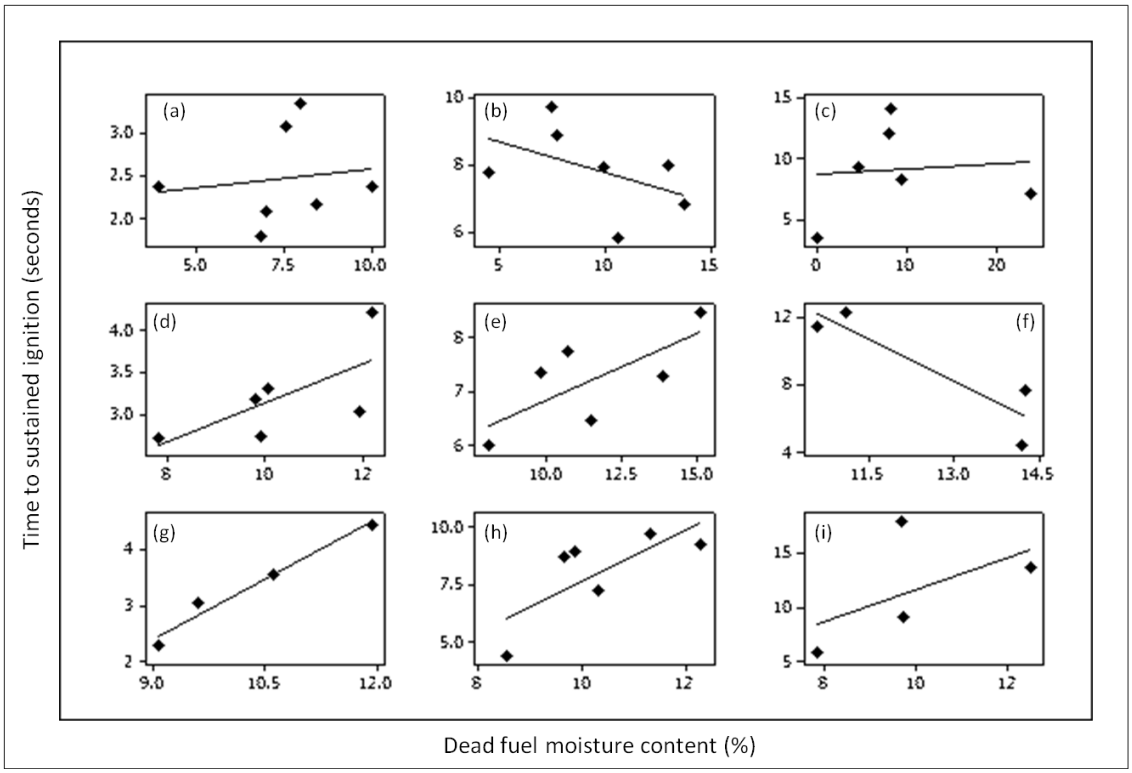

Figure 3.6 Relationship between time to sustained ignition of litter material and different litter moisture contents: (a) Callidendrous forest 0-3 mm; (b) Callidendrous forest 3-6 mm; (c) Callidendrous forest 6-25 mm; (d) Deciduous beech scrub 0-3 mm; (e) Deciduous beech scrub 3-6 mm; (f) Deciduous beech scrub 6-25 mm; (g) *Eucalyptus coccifera* woodland 0-3 mm; (h) *Eucalyptus coccifera* woodland 3-6 mm; (i) *Eucalyptus coccifera* woodland 6-25 mm.

Table 3.14 Regression equation, *P*-value and *R*-sq for the relationship between time to sustained ignition (TTSI) and dead fuel moisture content (DFMC) for *Eucalyptus coccifera* 0-3 mm diameter class.

Habitat	Diameter class	Equation	*P*-value	*R*-sq (%)
Eucalyptus coccifera (*n* = 6)	0-3 mm	TTSI = -4.016 + 0.7144 DFMC	0.017	0.97

Regression analysis of time to sustained ignition and DFMC showed only the 0-3 mm diameter class for the *Eucalyptus coccifera* woodland to be significant (Table 3.14; $P = 0.017$). There was no significant difference in the time to sustained ignition for fine fuels between vegetation types when the average time to sustained ignition for all moisture contents was examined (Kruskal-Wallis *H*-value = 1.76; *P*-value = 0.415).

Time to sustained ignition was significantly influenced by the litter size classes for each vegetation type (Figures 3.7 – 3.9). In all cases sustained ignition occurred fastest in the smallest litter size class. For the deciduous beech scrub and *Eucalyptus coccifera* woodland, sustained ignition was next fastest in the 3-6 mm size class, followed by the 6-25 mm class. Only for the callidendrous forest

was there no significant difference between the time to sustained ignition between the 3-6 mm and 6 -25 mm size classes (Figures 3.7 – 3.9).

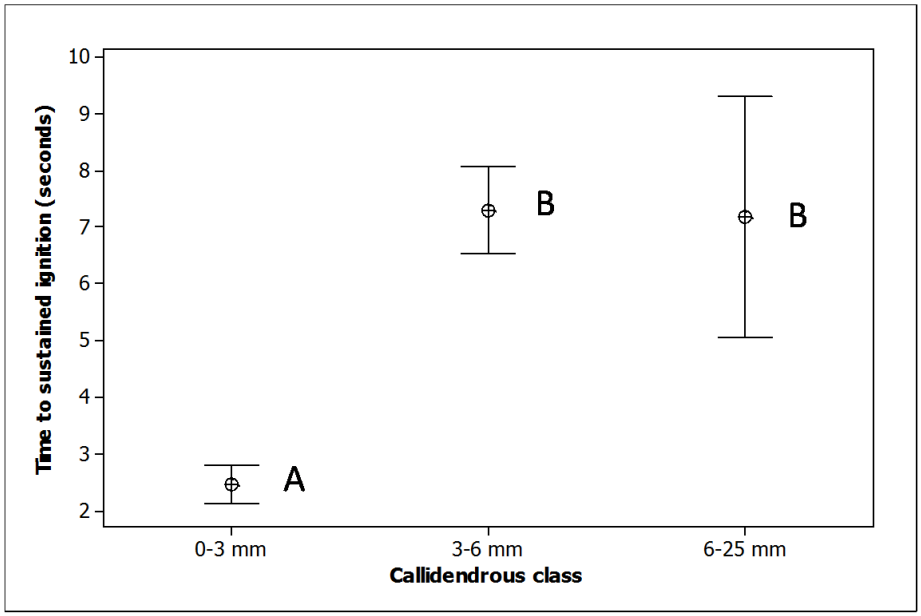

Figure 3.7 Interval plot of time to sustained ignition of litter and litter size class for callidendrous forest litter across all moisture contents (95% confidence intervals). Means that share a letter are not statistically different at P <0.05.

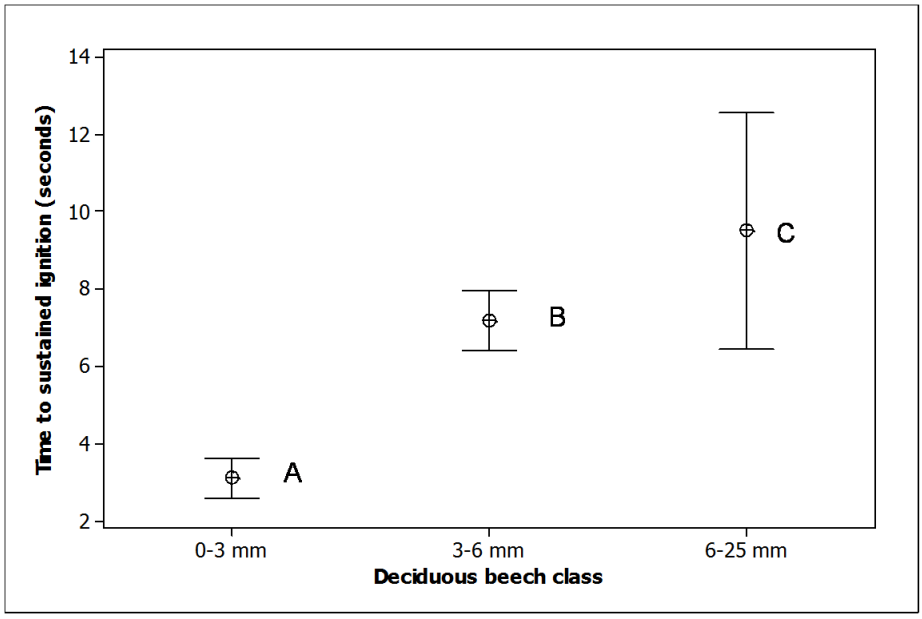

Figure 3.8 Interval plot of time to sustained ignition of litter and litter size class for deciduous beech scrub litter across all moisture contents (95% confidence intervals). Means that share a letter are not statistically different at P <0.05.

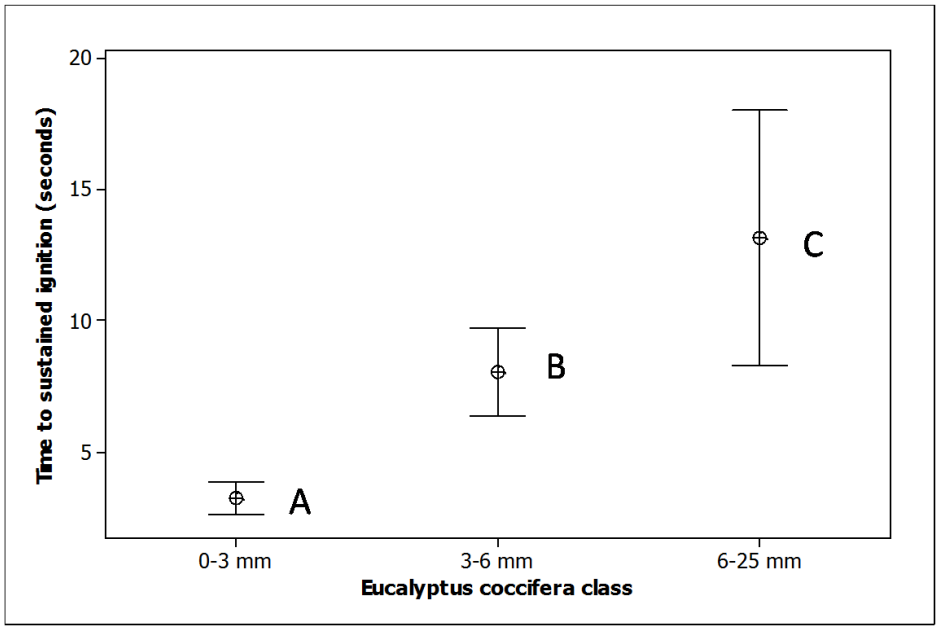

Figure 3.9 Interval plot of time to sustained ignition of litter and litter size class for *Eucalyptus coccifera* woodland litter across all moisture contents (95% confidence intervals). Means that share a letter are not statistically different at P <0.05.

3.3.3 Vegetation type

Sustained ignition for woody litter samples in each vegetation type was deemed to have occurred when 50% of the litter samples at each moisture content indicated continuing combustion. For the callidendrous forest sustained ignition was recorded for the 0-3 mm litter class at about 8% DFMC. The 3-6 mm litter class recorded a sustained ignition at about 14% DFMC, whereas for the 6-25 mm litter class, sustained ignition occurred at about 9% DFMC.

For the deciduous beech scrub the 0-3 mm litter class recorded a sustained ignition at about 10% DFMC. The 3-6 mm litter class recorded sustained ignition at about 15% DFMC, while the 6 -25 mm litter class recorded sustained ignition at about 14% DFMC.

The *Eucalyptus coccifera* woodland recorded a sustained ignition at about 9% DFMC for the 0-3 mm litter class. Sustained ignition was recorded at about 10% DFMC for the 3-6 mm litter class, whereas sustained ignition was recorded at about 12% DFMC for the 6 -25 mm litter class. The *Eucalyptus coccifera* woodland showed the greatest variation in sustainability of ignition within litter related to moisture content, with a higher moisture content frequently producing more instances of sustained ignition than the subsequent lower moisture content. Influence from volatile compounds within the samples was noted, with hissing and spluttering during burning, as well as

emissions of fierce puffs of smoke often capable of extinguishing the flame. Even after being oven dried, influence from volatile compounds was still evident in the smallest litter size class.

3.3.4 Field test fires

The best predictor of fuel load was leaf litter cover * height (R-sq = 0.67; P = 0.004). The relationship was explained by the following equation:

Fuel load = 0.620 + 0.00232 * (leaf litter cover * height)

Equation 3.3

Only 19 test fires were able to be carried out over the summer and autumn period of 2013, and of these only four fires managed to sustain ignition. The fires in which ignition was sustained were in elevated, more aerated fuel, rather than the densely compacted leaf and branch litter of the forest floor.

3.4 Discussion

Flammability can be assessed at a community level or at a species or individual plant level. The analyses undertaken in this study provide an indication of the relative flammability of rainforest and non-rainforest components of the respective communities. There are undoubtedly problems in extrapolating the results of the laboratory studies to entire communities as individual plant components are not always going to be representative of the entire plant or species. Another problem in extrapolating results from only a severed section of plant is that the natural processes that may occur in a plant under certain conditions (i.e. water stress) are altered. The water stressing of a single component does not allow for moisture transport within the whole plant and the natural conditions are thus altered (Etlinger and Beall 2004). However, leaves are less variable in their flammabilities than other parts of the plant (Gill and Moore 1996). The use of single leaves avoids the complication of comparisons with multiple leaves, and also avoids the lack of a direct measure of resistance to ignition involved with the use of finely ground material (Gill and Moore 1996; Etlinger and Beall 2004). However, the measurements of the flammability of *in situ*

rainforest litter gave similar values for the fuel dryness necessary to sustain fires as the laboratory measurements of rainforest leaves and litter.

Of the factors tested in the laboratory experiments, moisture content was consistently the single most important variable in determining the flammability of the individual plant components. Moisture content influenced the ignitability, sustainability, combustability and consumability of the specimens. This is consistent with the findings of many other authors (Gill and Moore 1996; Dimitrakopoulos and Papaioannou 2001; Behm *et al.* 2004; Etlinger and Beall 2004; Fernandes *et al.* 2008; Leonard 2009; Marino *et al.* 2010). Moisture acts as a heat sink, where heat is lost for fuel moisture to evaporate, diluting volatile oils as well as excluding oxygen from the combustion zone (Dimitrakopoulos and Papaioannou 2001; Etlinger and Beall 2004). The generalised linear model examining rate of spread (combustability) found that moisture content, species and the interactive effect of moisture content and species accounted for 57% of the variation in rate of spread among foliar leaf species. Similarly, the general linear model that best explained the variation in time to ignition (ignitability) found that moisture content and species explained 62% of this variation. This is, however, lower than the figure of Dimitrakopoulos and Papaioannou (2001), who found that moisture content alone was determining 73 – 94% of the variation in time to ignition of Mediterranean forest fuels, and Gill and Moore (1996), who found that moisture content and surface area to volume ratio explained over 80% of the variance in time to ignition of the leaves of some native Australian species.

The fine fuel litter from the three vegetation types examined in this study did not sustain ignition in the laboratory until moisture levels were between 8 and 14% for the callidendrous forest, 10 and 15% for the deciduous beech scrub and below 10% for the *Eucalyptus coccifera* woodland. These values are much lower than values obtained by Gillon *et al.* (1995) and Fernandes *et al.* (2008) for maritime pine forests in Portugual and France. The predominance of needles in the litter of these pine forests may explain their high moisture contents when sustaining fire, as the physical and chemical characteristics of pine needles makes them very flammable (Fernandes *et al.* 2008). The results of the field test fires tend to support the lower fuel moisture levels needed to sustain ignition in the laboratory experiments. Test fires only sustained ignition when the fuel moisture was 16% or lower, with 14% the lowest recorded fuel moisture level, on a dry weight basis. It should be noted, however, that none of the test fires in which ignition was sustained continued to burn with vigour, with all 'successful' test fires just managing to reach the 5 minute criterion before self-extinguishing. It is interesting that the fine fuel litter, excluding bark, from the *Eucalyptus coccifera* woodland exhibited lower flammability based on the sustainability factor

than the two rainforest communities. This is in contrast to the literature that suggests that eucalypts are especially adapted to propagate fire (see for example, Jackson 1968; Mount 1970; Gill 1997). The litter from the *Eucalyptus coccifera* woodland contained a lot of green wood. This was identified as being green from moisture appearing on the stick ahead of the flame front during burning. This appeared to reduce the moisture content level at which the community was deemed to be able to sustain fire as the green litter almost never sustained ignition. Green samples would expel moisture and fierce puffs of smoke that frequently extinguished the flame. However, in some species greenness does not appear to be a hindrance to burning. The 19th century explorer Giles noted the boughs of the mallee tree 'burn almost as well green as dry' (quoted in Gill 1997). The green litter should not be discounted as it was a normal component of the ground litter and thus represents the community. It is possible that the *Eucalyptus coccifera* woodland contained a greater amount of green wood than other eucalypt forests as it occupies the subalpine zone. Therefore, the large amount of green wood observed may be due to high winds and minimal shelter from surrounding vegetation that has caused live branches to be stripped from the trees. Also, different eucalypt species have different characteristics, which affect flammability (Gill 1997). As *E. coccifera* is a subalpine species (Williams and Potts 1996) it would not be surprising that it is less flammable than lowland forest species, as subalpine communities harbour many fire sensitive species, suggesting the fire frequencies in these areas are much lower than certain lowland communities, such as *E. obliqua* and *E. regnans* forests.

Another variable that has been found to influence the flammability of vegetation components is the surface area to volume ratio (Papió and Trabaud 1990; Gill and Moore 1996). The present study found a faster rate of spread for litter samples that had higher surface area to volume ratios (SA/V) in all vegetation types. This is consistent with other studies (Papió and Trabaud 1990). The results for the foliar samples are harder to interpret. Across all fuel moistures only *Eucryphia lucida* and *Eucalyptus nitida* recorded significant relationships between rate of spread and SA/V. *Eucryphia lucida* and *Eucalyptus nitida* were the most combustible of the leaf species studied, based on their rate of spread, however sit at opposite ends of the SA/V spectrum, with *E. nitida* recording the lowest SA/V and *E. lucida* recording the highest. When the foliar species were examined as a group at respective drying times, the 72 hour drying treatment produced a significant relationship between SA/V and rate of spread however neither the 120 hour or oven-dried treatments were significant. Seventy-two hours was the first drying time in which all species recorded any rate of spread, and relationships at shorter drying times were influenced by the large number of no results. It is likely that the moisture contents of the species at the 120 hour

and oven-dried times are so low that there is little differentiation in the rate of spread, regardless of the SA/V.

Bauera rubioides appeared to show the least propensity to flammability in the laboratory, yet is found in every habitat from moorland to rainforest. The low flammability of *B. rubioides* is attributed to the stem, as individual leaves were not tested due to their insufficient size. This makes the result for *B. rubioides* not comparable with the other foliar specimens, but nonetheless represents the natural state of *B. rubioides* in the environment.

The response of *B. marginata* in this study was very different to that observed by Dickinson and Kirkpatrick (1985), who found *B. marginata* to self-propagate the flame and also produce explosive crackles and sparks. This may be a result of environmental differences in the growing conditions of the specimen trees. Gill and Moore (1996) state that measuring the flammability of a plant or species is difficult due to variations with life stage as well as management history. The *B. marginata* specimens collected by Dickinson and Kirkpatrick (1985) were from south-eastern Tasmania, an area that receives a mean annual rainfall of between 800 and 1,000 mm, compared to south-western Tasmania, where the mean annual rainfall is between 2,400 and 3,000 mm (Bureau of Meteorology 2012*b*). Therefore, it may be that differences in the environment and management history of the specimens produce such varied responses. It may also be due to intra-specific differences within *B. marginata*, which is widely distributed in Tasmania and highly variable in form (Blake and Hill 1996). It is possible that this variability also extends to its flammable properties. Another possibility is the different life stages that the sample trees were at when specimens were collected. There is likely to be variation in the flammability of the same species at the seedling, sapling, mature and senescent life stages (Gill and Moore 1996).

Nothofagus gunnii demonstrated a broad response to the laboratory flammability studies. This was probably due to the position of the leaves on the branches, as at higher moisture contents the leaves often carried flame but the flame was unable to spread along the branches to the next leaf unless the next leaf was close by. The branches required a much lower moisture content than the leaves before the flame would propagate along them. It is therefore possible that the low flammability levels observed in deciduous beech communities (Pyrke and Marsden-Smedley 2005) are partly a function of the structure of the vegetation. *Eucryphia lucida* leaves displayed a high flammability after short periods of drying. This is probably due to the thinness of the leaves, resulting in very rapid moisture loss.

The laboratory experiments indicated that there appeared to be some inconsistencies between the flammability of the leaves of each species and flammability of the litter in the vegetation types they occupy. *Atherosperma moschatum* leaves were very resistant to flame, however the litter from this species appeared very flammable, and lead to a much higher level of flammability for the callidendrous forest than if the litter had consisted of *N. cunninghamii* branches alone. *Eucalyptus nitida* leaves did not take much drying before achieving rapid combustion and ignition, represented by the rate of spread and time to sustained ignition. However, the branch litter from *E. coccifera* (which is a cline form of *E. nitida* (Williams and Potts 1996)) took a large amount of drying before it reached a level of flammability that could be deemed sustainable. In contrast, the bark litter that was present in the *Eucalyptus coccifera* woodland recorded a significantly greater rate of spread than any of the other litter classes across all three vegetation types. It is therefore likely that it is the bark litter that is responsible for the high flammability of this eucalypt community, consistent with the importance of bark in other eucalypt communities (Mount 1970; Gill 1997).

Although four test fires managed to achieve the sustainability definition of remaining alight for more than 5 minutes, none of these fires achieved this with much vigour, even though rain had been excluded from the plots for in excess of 83 days. Uhl *et al.* (1988) found similar results, as they were unsuccessful in starting experimental fires in a tall closed canopy rainforest in the Amazon when rain was artificially excluded for 41 days. Uhl *et al.* (1988) found this to be a result of the relative humidity remaining high under the rainforest canopy. They were, however, successful in lighting fires in nearby short-statured and open-canopied forest when the relative humidity dropped below 65% (Uhl *et al.* 1988). As the drying of fuel is dependent on exposure to wind, solar radiation and a low humidity (Marsden-Smedley 2009), it is not surprising that fire was able to sustain in the more open-canopied forest. In the present study, the near atmosphere within the forest was calm and although the air temperature rose during the day, the relative humidity did not drop below 90%. Perhaps if the test fires had been conducted earlier in the season, when the humidity within the forest had a chance of being lower, more experimental fires may have been sustained.

The structure of the fuel bed within the rainforest may be another factor contributing to the low flammability observed in the field experiments. Fernandes *et al.* (2008) found that fuel moisture content is influenced by the fuel layer structure, as less compacted fuels raise the moisture content threshold for fire propagation. Similarly, Uhl *et al.* (1988) found the compactness of fine fuels within a tropical rainforest negatively affected the chance of fire burning successfully. The fires that were able to sustain ignition were those that occurred in areas where the surface fuel

layer was elevated. Gould *et al.* (2007) have defined this as the near-surface fuel layer, and found that the product of the cover of the near-surface fuel layer by the height of the near-surface fuel layer is the best predictor of rate of fire spread.

The results of the present study indicate that there is no great inherent difference in the flammability of leaves and litter between species that dominate rainforest and species that occupy the high fire frequency communities that occur adjacent to the rainforest in south-western Tasmania. This is in contrast to other studies that show a greater flammability of species from more fire prone environments (Kirkpatrick and Dickinson 1985; Behm *et al.* 2004). The rainforest microclimate, the observed structure of the fuel bed within rainforest communities and the absence of strips of decorticating bark appear to contribute to a low flammability rather than any properties of the species themselves, or of their leaf and small branch litter. Furthermore, the results of the relationships between SA/V and flammability properties indicate that SA/V cannot be used to extrapolate the flammability results of other species within the two broad forest ecosystems which were examined.

Chapter 4 – Structure of the rainforest canopy

4.1 Introduction

Forest canopies greatly reduce the amount of precipitation reaching the forest floor, which affects water storage in the soil profile (Link *et al.* 2004; Staelens *et al.* 2006). This interception may result in the soil dryness within a forest being greater than the soil dryness outside of the forest. Thus, interception can increase the flammability of the litter layer and understorey of a forest (Ray *et al.* 2010).

In order to better understand the relationship between precipitation and flammability within a forest it is necessary to determine the amount of precipitation that is intercepted by the canopy, the amount of precipitation that can be held within the canopy before throughfall occurs, and develop an understanding of how a canopy effects the spatial distribution of rainfall at ground level.

Canopy interception (I) is defined as the part of the rain that falls on vegetation and evaporates without reaching the ground (Klaassen *et al.* 1998). I cannot be measured directly but must be measured from rainfall above the forest canopy less rainfall that reaches the forest floor. I is composed of the amount of water held in the canopy at any given time (C) and evaporation from the canopy surface during the rain event (E). Not all rain events will deliver sufficient water to satisfy maximum values of C whereas others greatly exceed this amount (Dunkerley 2000). When this occurs C becomes S as canopy saturation capacity is reached and water begins to drip from the leaves, reaching ground level, known as throughfall (Tf). Tf is composed of three components. Free throughfall (T_f) is rain that reaches the forest floor without striking any vegetation. Released throughfall (T_r) is rain that is released from the canopy once S is reached. The third component of Tf is stemflow (S_f), water that is funnelled to the ground via the stem. Some stem and bark storage may occur with this mode of water transport.

I can therefore be expressed as:

$$I = P_g - Tf - E$$

Equation 4.1

where Pg is gross rainfall and

$$Tf = T_f + T_r + S_f$$

Equation 4.2

Table 4.1 Abbreviations associated with canopy interception that occur in Chapter 4.

C	Canopy storage. The volume of water that is held in the canopy.
I	Canopy interception loss. Canopy storage plus evaporation.
S	Canopy saturation value. The amount of water required to wet the leaves only. S is a discrete point in time that occurs once C is reached and as the T_r and S_f components of Tf begin.
P_g	Gross rainfall.
Tf	Throughfall. Composed of T_f, T_r and S_f.
T_f	Free throughfall. Rain that falls through the canopy without striking any vegetation.
T_r	Released throughfall. Rain that falls through the canopy after being released once C is reached.
S_f	Stemflow. The water that reaches the forest floor by flowing down the stems of vegetation.
E	Evaporation from the forest canopy.

Canopy interception (I) and saturation (S) values have been determined for a variety of climatic regions and forest types, including tropical rainforest (Jackson 1975; Herwitz 1985; Germer *et al.* 2005), temperate conifer forest (Klaassen *et al.* 1998), temperate rainforest (Link *et al.* 2004), temperate conifer plantation forest (Bell and Gatenby 1969; Rutter *et al.* 1972) and temperate eucalypt forest (Bell and Gatenby 1969; Crockford and Richardson 1990). All authors found considerable variation in the values for canopy interception parameters, which is not surprising considering the diversity of forest types and climates examined. However, considerable variation was also common among forests broadly grouped within the same class. For example, Rutter *et al.* (1975) modelled the canopy saturation capacity of a Douglas-fir forest in England and found it to be equivalent to 1.2 mm of rainfall, whereas Klaassen *et al.* (1998) found the same parameter to be double that of the English forest in a Douglas-fir forest in The Netherlands. Furthermore, Link *et al.* (2004) recorded a value of 3.3 mm in an old growth Douglas-fir – western hemlock forest in the Pacific North West of the USA. The considerable variation in these figures indicates that it is not possible to extrapolate values within and across respective forests and climates (Jackson 1975; Herwitz 1985; Klaassen *et al.* 1998; Crockford and Richardson 2000; Dunkerley 2000), unless the precipitation characteristics are identical (Jackson 1975). Therefore, in order to determine canopy intercept parameters for rainforest in southern and western Tasmania it is necessary to conduct canopy interception studies for the particular sites and forest types of interest.

Past studies of canopy intercept parameters have used a number of different methods to measure *I* and *S*. These include microwave transmission (Klaassen *et al.* 1998), water soaking (Herwitz 1985; Crockford and Richardson 1990), artificial rain (Aston 1979; Herwitz 1985), modelling (Rutter *et al.* 1972; Gash 1979) and measurements of direct throughfall (Jackson 1975; Link *et al.* 2004; Germer *et al.* 2005).

In naturally occurring forests, estimates of *I* and *S* are influenced by a high spatial variability of *Tf* (Jackson 1975; Hutjes *et al.* 1990; Bellot and Escarre 1998; Germer *et al.* 2005; Staelens *et al.* 2006; Zimmermann *et al.* 2010). This is caused by concentrated areas of *Tf*, such as drip points (Zimmermann *et al.* 2010) as well as differences in the Leaf Area Indices (LAI) and canopy cover within the forest (Germer *et al.* 2005). LAI has explained the spatial variation of *Tf* in some studies (Ford and Deans 1978; Johnson 1990) but not others (Bellot and Escarre 1998; Carlyle-Moses *et al.* 2004). Carlyle-Moses *et al.* (2004) attributed this to the relatively high LAI in the studies of Ford and Deans (1978) and Johnson (1990), whereas the LAI of Bellot and Escarre (1998) and Carlyle-Moses *et al.* (2004) were much lower. Canopy projective cover (CPC) has also been used to explain *Tf* variation in some studies (Bellot and Escarre 1998; Staelens *et al.* 2006).

The type of rainfall falling on a forest canopy varies over time in response to variation in direction, intensity, continuity, and rain drop size (Crockford and Richardson 2000; Germer *et al.* 2005). This variation, along with other variables such as wind speed, has been shown to result in differing values of *I* between events of similar *Pg* (Crockford and Richardson 2000).

S will be affected by variables such as wind speed and evaporation (*E*). Under high winds the resultant wind shake will significantly reduce the amount of water that can be held in the canopy before either T_r or S_f begin (Jackson 1975). Herwitz (1985) found this value to be as much as 50% of the total volume of water detained in still air. However, the factor of wind speed is almost always ignored (Dunkerley 2000). The rate of *E* will be affected by time of year, the nature of the rainfall (continuous or intermittent), the amount of water held on the canopy, and the humidity deficit (Klaassen *et al.* 1998). It is assumed that *E* is the dominant force on *I* once rain has ceased. However, Klaassen *et al.* (1998) has shown that *C* is the dominant component of *I* during a rain event, with *E* being of minor importance. As the present study is primarily interested in water contribution to the soil profile a quantification of *E* is not of importance and *E* consequently becomes a component of *I*.

S_f can be a significant contributor to *Tf*, with some estimates being as high as 13% of the total *Tf* (Crockford and Richardson 2000). Stems also have a great potential for water holding capacity.

Herwitz (1985) found that woody surfaces on branches and tree trunks can contribute over 50% of the total water held by the tree in still conditions and this figure can rise to as high as 80% under windy conditions. This value is highest for tree species that have a thick, rough bark and presumably also for species that have large numbers of epiphytic mosses and lichens (Link *et al.* 2004).

The present study is primarily directed towards the contribution that precipitation makes to the soil and fuel dryness within a forest. Therefore, only water that reaches the ground is of interest and the water holding capacity of the trunk is irrelevant, provided that there is no downward movement of this water to the soil profile. The large numbers of epiphytic mosses and lichens found on many of the trees within cool-temperate rainforest will assist in retaining water above the soil profile.

Some species are more conducive to S_f then others. This variation relates to their leaf and branch characteristics. Crockford and Richardson (2000) state that leaves that are angled above the horizontal, i.e. the leaf tip is above the petiole, and have a concave shape, will have higher S_f. Also, species with steep branches have a greater potential for S_f than species with horizontal or below horizontal branches (Herwitz 1986), with the latter making no contribution to S_f (Crockford and Richardson 2000). As the dominant tree species in cool-temperate rainforest tend to hold their branches close to the horizontal, and have either downward facing (*Atherosperma moschatum*) or horizontal to downward facing (*Nothofagus cunninghamii*) leaves, S_f has not been examined in the present study. This is further justified by the very small S_f values that have been obtained in structurally similar vegetation types from other regions (Crockford and Richardson 2000).

This chapter will focus on the determination of values of I and S for a callidendrous rainforest as influenced by wind direction, wind speed and wet or dry canopy events. A second objective of this chapter is to examine the distribution of rain water within the rainforest and to determine the factors influencing this variation.

4.2 Methods

4.2.1 Site selection

Canopy interception parameters were measured at Site Call (Chapter 2). The understorey of Site Imp was too tangled to be conducive for the placement of large amounts of equipment, while Site DB also suffered from a tangled understorey but was also too small in area to provide an effective result. As the site required frequent visiting for the duration of the canopy intercept study, Site Org, which was more distant, was also impractical. Furthermore, Site Call exemplified a standard callidendrous rainforest. Callidendrous rainforests are more uniform than implicate rainforests, where the structure can vary greatly. The derivation of canopy intercept parameters for a callidendrous rainforest site would therefore provide the greatest application for Tasmanian cool-temperate rainforests generally.

4.2.2 Distribution of rain gauges

Site Call was divided into a 50 x 50 m grid, with each grid square being 10 x 10 m. At the extremities of the grid, some of the 10 x 10 m squares lay in forest that was classified as thamnic rainforest. Where this occurred, that grid square was excluded and an additional grid square was placed adjacent to the 50 x 50 m grid, in callidendrous rainforest. Within each grid square one Nylex 1000® funnel rain gauge was established using randomly generated co-ordinates, resulting in 25 rain gauges in total. Each gauge had a collecting area of 86.6 cm^2 and was placed with its rim 1.3 m above the ground.

4.2.3 Data collection

The rain gauges were emptied after each rain event, or as soon thereafter as practicable, between 2 September 2010 and 22 February 2011 and between 20 April 2012 and 16 May 2012. Some of the data collected between these dates were discarded from analysis. Instances in which a rain-free interval exceeded 24 hours were classified as intermittent rain events. This is in contrast to continuous events, in which the rain-free intervals did not exceed 24 hours. When less than 24 hours elapsed after the emptying of the rain gauges and the onset of further rain this was classified as a wet canopy event.

A tipping bucket rain gauge was established at the Control 1 station (Chapter 2). This site was chosen as it was the closest open area suitable for the establishment of an automatic weather station. The distance between the sites was three kilometres. In order to mitigate a potential problem with this distance two additional funnel rain gauges were established with one being next to the tipping bucket and the other located above a road cutting about 500 m from the rain gauges at Site Call. Additional collection from a tipping bucket rain gauge located under the canopy at Site Call and placed roughly in the centre of the 50 x 50 m grid was made during the second collecting period, from 20 April 2012 to 16 May 2012. Additional weather data as detailed in Chapter 2 were recorded during both of the rain collection periods. No weather data from the control are available for Event 2.

Hemispherical canopy images were taken at the locations of all the funnel rain gauges in fine weather conditions during autumn 2011. The photos were taken by removing the rain gauges and placing the camera lens in the same vertical plane as the top of the gauge. The canopy images were taken facing upwards with a Nikon Coolpix 950 camera with an EK80 fisheye lens attached to horizontally supported gymbals. The supporting frame includes an internal magnetic compass and north/south markers so that orientation of the image is captured in each photograph. All photos were taken in diffuse skylight conditions, either in heavy overcast periods or at dusk. This avoids problems where parts of the visible sky may be darker than the canopy (e.g. a dark cloud) or parts of the canopy are brighter than the visible sky (e.g. reflections of sunlight on leaves; Delta-T Devices 1999). Each image was then classified according to black (representing cover) and white (open sky) components using a threshold algorithm which is included in the HemiView version 2.4 program (Delta-T devices 1999). At least three replicate photographs were taken at each rain gauge location. The rain gauges were returned to their positions after the photographs had been completed. All trees within a 5 m radius of each rain gauge were recorded for species, diameter at breast height (DBH), direction and distance from the gauge.

4.2.4 Data analysis

The Leaf Area Index (LAI) of the forest canopy as observed from each rain gauge site was determined using the HemiView software program (Delta-T Devices 1999). HemiView estimates the effective single-sided LAI. Since trunks and branches are included in this estimate the recorded value for LAI will be referred to here as Canopy Area Index (CAI). CAI and canopy

projective cover (CPC) were calculated as the average of the individual estimates produced for three or more replicate photos taken at each rain gauge location. For each hemispherical image, gap fraction (GF) was determined based on the aggregate proportions of white vs. black in each of 112 radial grid cells overlain per image. Percentage CPC was then derived for each image as the value of (1 – GF) x 100. The standard deviations for CAI and CPC were then determined for each rain gauge location.

Regression equations were derived to examine the relationship between the rain collected in the funnel rain gauge placed at the control station (y) and the gauge placed nearby the sample site. The equation was highly significant with an R-sq value of 0.99. The best-fit regression equation was:

$$y = 0.9187x - 1.5457$$

Equation 4.3

Similarly, the relationship between the tipping bucket located under the canopy at Site Call (y) and the average of the funnel rain gauges within the grid was assessed using linear regression. Once again, the relationship was found to be highly significant with an R-sq value of 0.99. The equation that explained the best-fit was:

$$y = 1.1265x + 0.5978$$

Equation 4.4

The raw data collected from the control station were adjusted using equation 4.3, in order to mitigate the effect of distance between the sites. Any values for rain events that became negative after adjustment were removed from analysis. The data from the tipping bucket within Site Call were adjusted using equation 4.4.

The rain events sampled during the second collection period (Events 7-11) were split into sub-events, with any gap in rainfall that was greater than two hours being the cut-off point for a new sub-event. This resulted in the creation of 15 sub-events. The adjusted data from the tipping bucket located within Site Call was then used as a proxy for rainfall below the canopy for the entire site during each of these 15 sub-events.

The duration, rainfall rate, mean wind direction, mean wind speed and wind speed range were calculated for each event and sub-event after removing gaps in rainfall greater than two hours.

The mean *Tf*, standard deviation and co-efficient of variance was calculated for the funnel rain gauge. Pearson's product moment correlation coefficient was used to test relationships between mean wind speed, maximum wind speed and mean wind direction and *I* as a percentage of *Pg* for all events that had associated wind data.

Line graphs were produced of an additional 12 rain events between 25 January and 16 April 2012, as well as Events 7 and 8. All events had dry canopies at the beginning of the first rainfall, with the exception of Event 8 and the event to occur between 6 – 11 April 2012. Graphs were constructed of the cumulative rainfall for each event as well as the total rainfall for each hour. These graphs were analysed to determine the patterns between *Tf* and *Pg* and the raw data were used to determine *S*, by taking the mean of the amount of *Pg* that fell before *Tf* commenced, for all events that had appropriate resolution of the data. This value was then adjusted using equation 4.4 to obtain the value for the whole study site.

Wind speeds were converted to km/hr at a height of 10 m above the ground by increasing the two metre value by one third (Marsden-Smedley *et al.* 1999). This procedure adjusts values to those measured by the Bureau of Meteorology, which places wind sensors at 10 m above the ground. The diameter at breast height (DBH) of the trees in the 5 m radius around each rain gauge were converted to basal area in m^2/ha.

A map was constructed using ArcMap version 9.3 of the distribution, size and species of trees around each of the funnel rain gauges. The hemispherical canopy images taken at the position of each rain gauge were used to calculate the distribution of canopy cover in an arc around the mean wind direction for each event, with the standard deviation around the mean used to determine the size of the arc. The arc was further reduced to exclude the outer 22% of its radius, in order to avoid the excessive distortion that is inherent within these outer values for any image taken with a fisheye lens. The exclusion of the horizon was further justified as canopy cover detail of trees on the horizon is not likely to be of relevance for determining *Tf*. Each photograph was converted to a black and white image and software written with Interactive Data Language (IDL) was used to count the number of black pixels (canopy cover) within the arc.

I was standardised for each event by subtracting the mean and dividing by the standard deviation (Minitab Inc 2010). The standardised *I* was then plotted against the percentage canopy cover for the arc around the mean wind direction for 10 of the recorded rain events.

Regression analyses were used to determine if there were any relationships between the CPC, CAI, number of trees within the 5 m radius or basal area for each rain gauge and the *Tf* collected for each rain gauge. Regression analyses were used to test for relationships between *I* as a percentage of *Pg* and *Pg*, *Tf* and *Pg* and *I* and *Pg*.

4.3 Results

A summary of each of the rain events including start date, gross rainfall, the type of event, the canopy status (wet/dry) at the start of the event, the duration of the event, the rainfall rate, the average wind direction and speed, the wind range, the total throughfall and the interception is presented in Table 4.2.

Table 4.2 Rainfall event number, start date of each event, gross rainfall (*Pg*), event type, (I = intermittent, C = continuous), canopy status at the start of each event (D = dry canopy, W = wet canopy), the duration of each event in hours, rainfall rate, mean wind direction, mean wind speed, wind speed range, throughfall (*Tf*), throughfall as a percentage of gross rainfall, interception (*I*), and interception as a percentage of gross precipitation.

Event	Start date	Pg (mm)	Event type	Canopy	Duration of event (hr)*	Pg rate (mm/hr*)	Mean wind dir (°)*†	Mean wind speed (m/s)* †	Wind range (m/s*†)	Tf (mm)	Tf as % of Pg	I (mm)	I as % of Pg
1	2/09/10	45.49	I	D	34	1.34	331	0.96	2.122	47.22	103.80	-1.73	-3.80
2	10/09/10	137.36	C	n/a	174	0.79	n/a	n/a	n/a	99.52	72.45	37.84	27.55
3	9/12/10	56.52	C	D	65	0.87	236	2.45	4.212	38.26	67.70	18.26	32.30
4	16/12/10	62.03	C	D	107	0.58	232	1.99	4.722	43.04	69.39	19.00	30.61
5	6/02/11	35.02	I	D	111	0.32	236	1.44	4.63	28.08	80.19	6.94	19.81
6	17/02/11	60.01	I	D	50	1.20	225	2.5	4.983	50.52	84.19	9.49	15.81
7	20/04/12	4.33	C	D	6	0.72	210	0.22	0.565	3.85	88.83	0.48	11.17
8	22/04/12	13.90	C	W	30	0.46	220	0.82	1.773	8.89	64.01	5.00	35.99
9	2/05/12	14.44	C	W	19	0.76	210	0.68	1.379	13.46	93.22	0.98	6.78
10	5/05/12	16.28	C	W	38	0.43	235	1.06	2.024	25.70	157.89	-9.42	-57.89
11	8/05/12	88.12	I	W	92	0.96	247	1.57	3.46	62.17	70.55	25.95	29.45

* excludes data relating to gaps in rain events greater than two hours
† wind data taken from two metres above ground level
n/a - no data available for that period

4.3.1 Temporal variation

The total rainfall that was collected at the control station during the study period was 533 mm. This occurred across 11 rain events, which consisted of four intermittent and 7 continuous events. There were four events in which the rainfall was less than 20 mm (Table 4.2). Events 7-11 were split into sub-events. This created a total of 26 events and sub-events. Of these, 18 recorded rainfall less than 20 mm, with 13 sub-events recording falls less than 5 mm (Figure. 4.1).

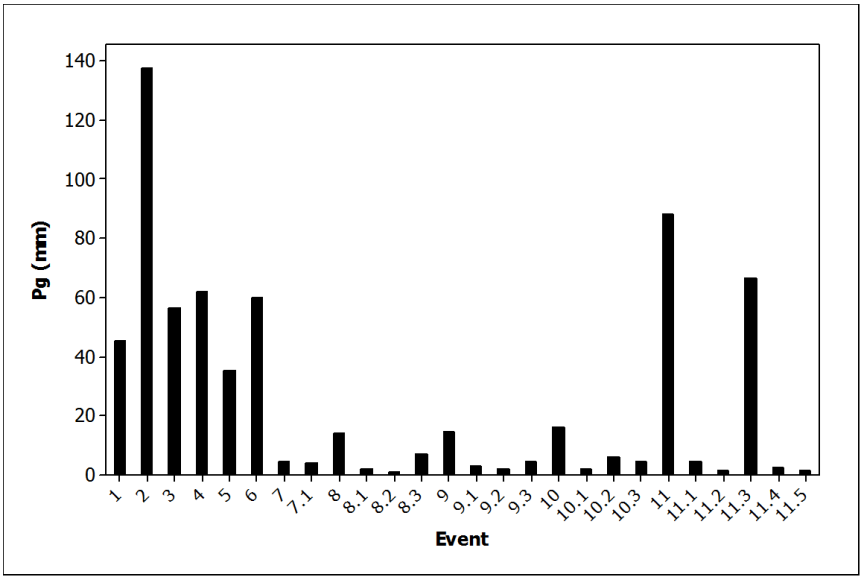

Figure 4.1 Size of rainfall events and sub-events.

The rate of rainfall for the events ranged from 0.32 mm/hr to 1.34 mm/hr (Figure 4.2). The average rainfall rate for the events was 0.77 mm/hr.

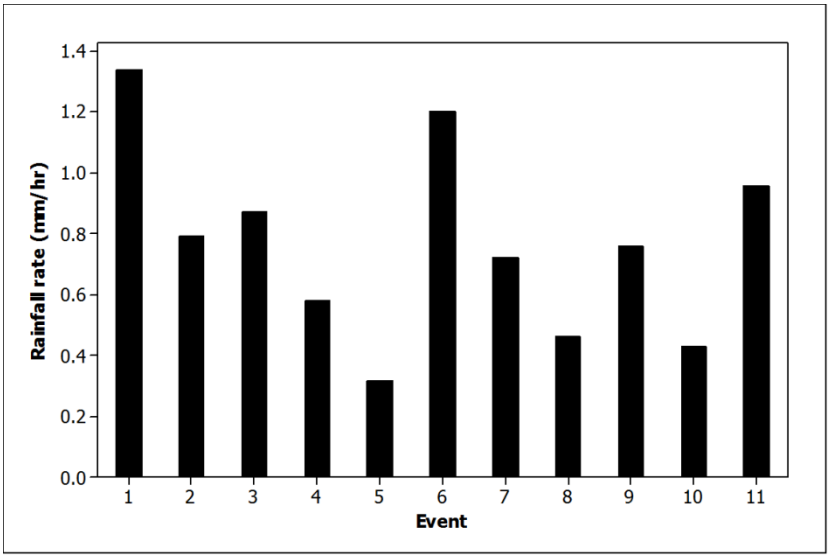

Figure 4.2 Rate of rainfall (mm/hr) for each of the 11 rainfall events.

Sub-event 8.3 recorded a much higher rainfall rate than any of the other events or sub-events, at 6.91 mm/hr. The lowest rainfall rate recorded for the sub-events was 0.20 mm/hr (Figure 4.3). The average rainfall rate for the events and sub-events was 0.99 mm/hr.

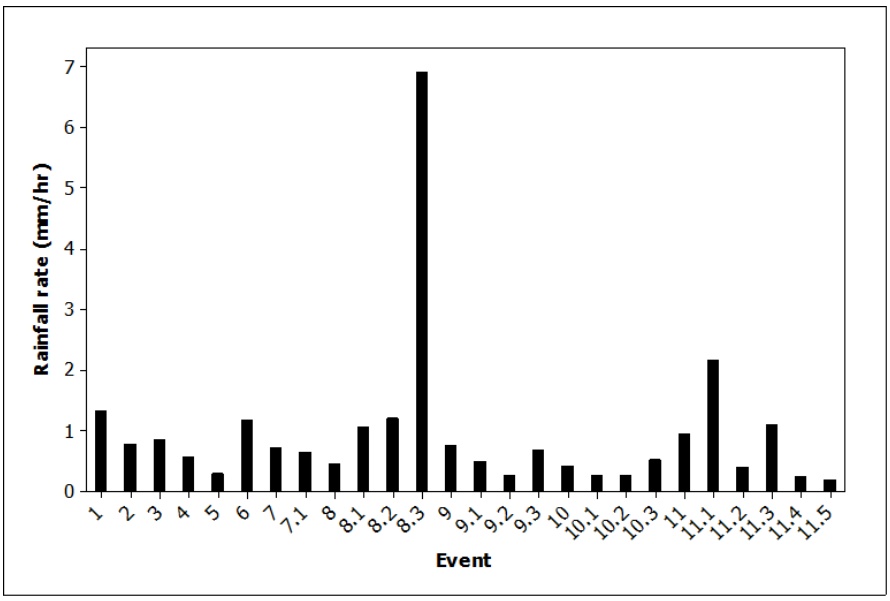

Figure 4.3 Rate of rainfall (mm/hr) of events and sub-events.

The weather conditions that occurred during the study period were not extreme. The highest recorded mean wind speed was 2.5 m/s or 12 km/h converted for a height of 10 m, during Event 6. The most extreme wind gust recorded at the control station was 4.98 m/s or 23.92

km/h converted for a height of 10 m, also for Event 6 (Table 4.2). There were no significant relationships between any of the wind variables and *I* as a percentage of *Pg*.

There appeared to be differences in the behaviour of the relationship between the *Tf* and *Pg* for small events (less than 8 mm) and large events (greater than 25 mm; Figure 4.4). During small events, *Tf* failed to register small falls that were recorded in *Pg*. In larger falls within the small events (defined by rainfall less than 8 mm) there tended to be a delay, and subsequent lag in *Tf*. For the large events the delay in *Tf* was not always so pronounced. During spikes in rainfall, *Tf* was often less than *Pg* (Figure 4.4).

For the dry canopy events, the difference between *Pg* and *Tf* as a percentage of the *Pg* tended to be large (50 – 81%) during small events and small (0.7 – 29%) during large events. For the event between 6 April and 11 April 2012 *Tf* was larger than *Pg* (Figure 4.4).

In 5 of the dry canopy events it took between 1.2 mm and 2 mm of *Pg* before *Tf* was registered. These figures correspond to the saturation value, *S*. In the other 6 events the resolution of the data did not enable this figure to be determined. A saturation value of 1.2 mm and 2 mm of gross rainfall were recorded once each and 1.4 mm three times. For the wet canopy event *Tf* started in the same hour as *Pg* (Figure 4.4).

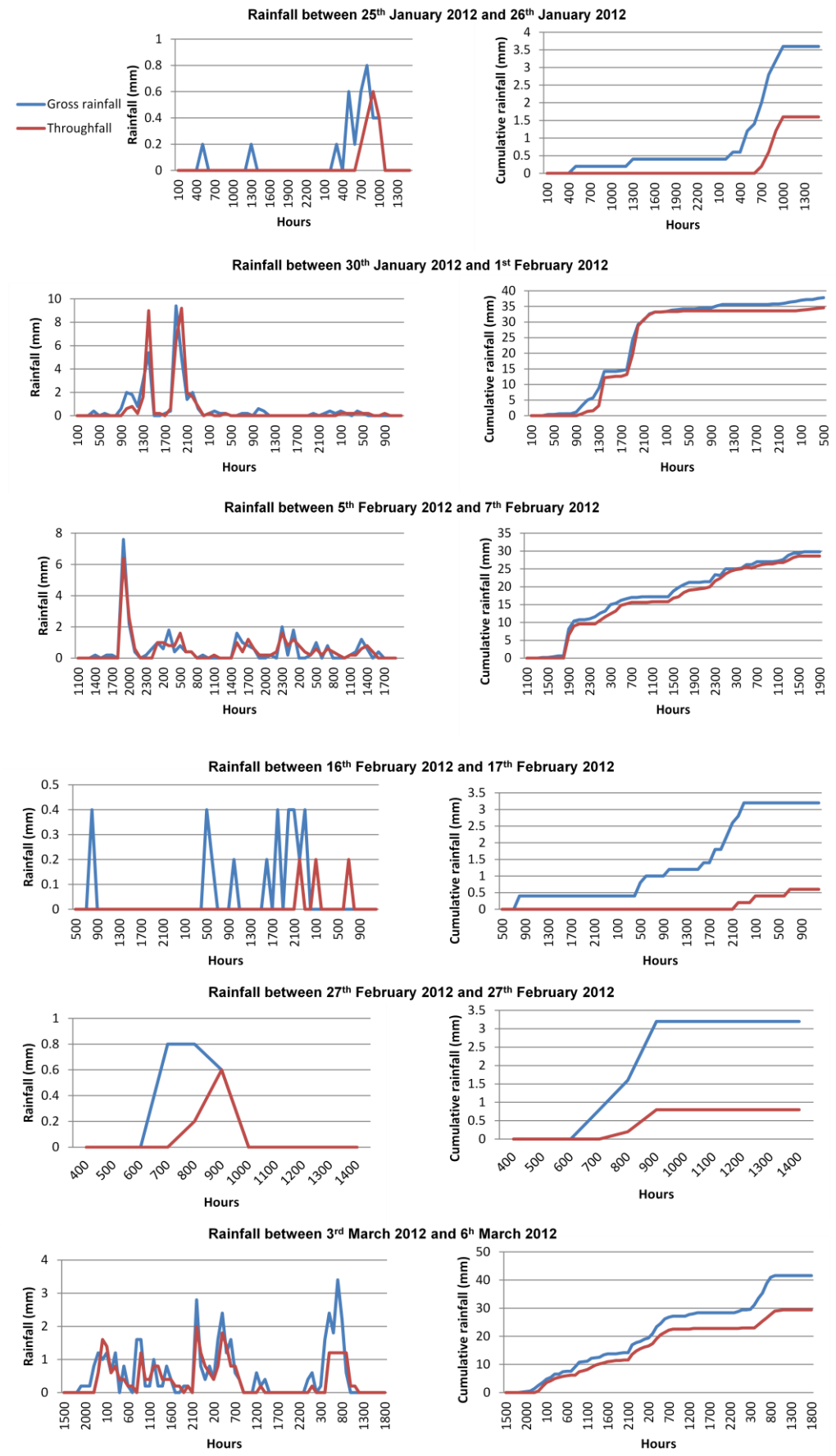

Figure 4.4 Total and cumulative gross rainfall and throughfall for separate events over time.

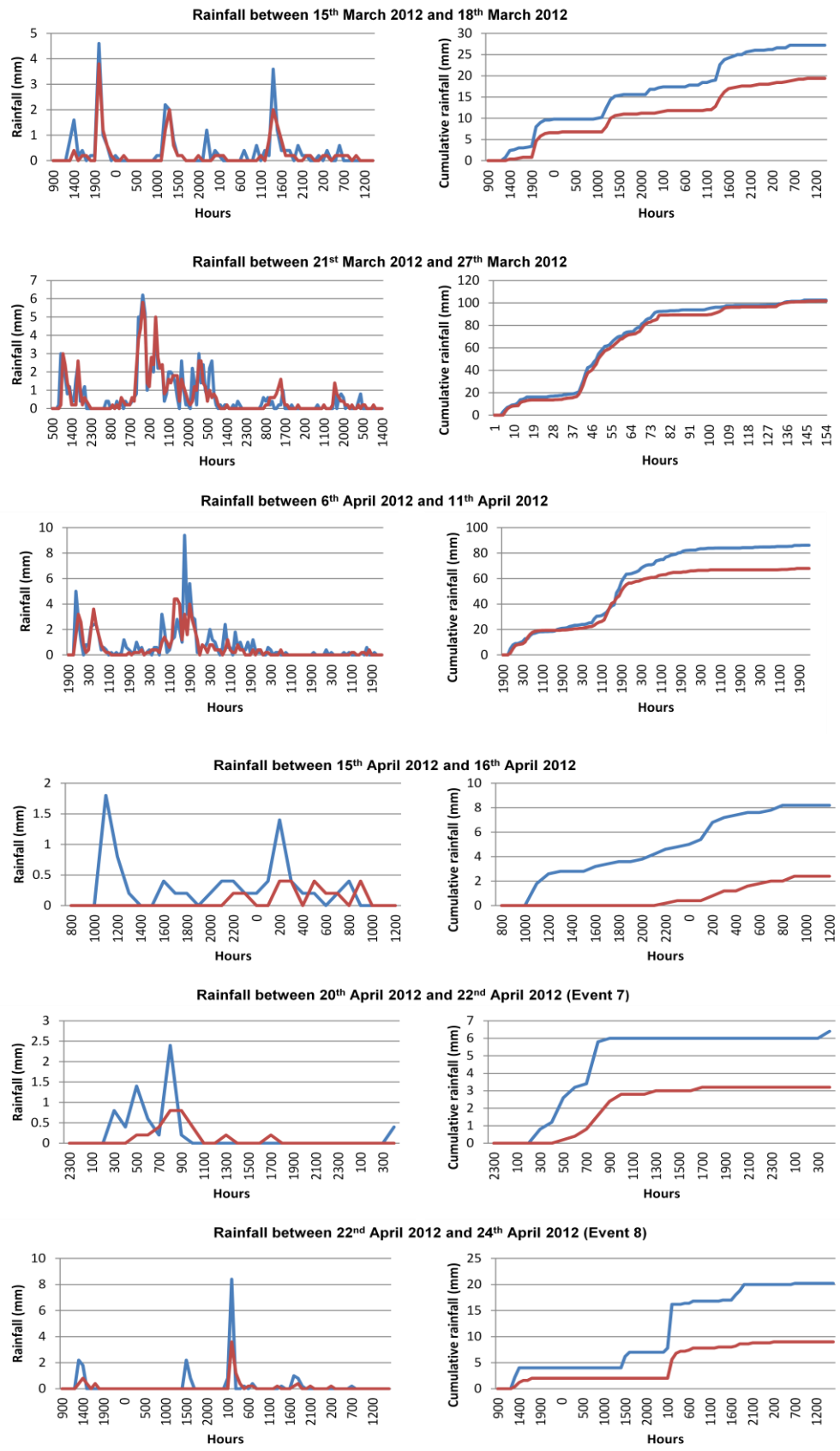

Figure 4.4 (continued). Total and cumulative gross rainfall and throughfall for separate events over time.

4.3.2 Spatial analysis

The mean *Tf* for the site across all 11 events was 425.5 mm. Nine of the 22 gauges that had a complete rain record over the entire study period show a cumulative deviation below the site mean. Of these, gauge 19 shows the most deviation in either direction at 265 mm below the site mean. Gauge 17 is the closest gauge to the site mean, with a cumulative deviation only two mm above the site mean, while gauges 16 and 25 show the most deviation above the site mean (Table 4.3).

The CPC above each rain gauge ranged from 87% at gauge 8 to 95.7% at gauge 15, with a mean of 91.8% across the site, while the mean CAI was 3.1 (Table 4.3). The mean basal area for the site was 116.8 m^2/ha. The mean number of trees in the 5 m radius around each rain gauge was 9.8, ranging from three, at gauge 14 to 18 at gauge 13 (Table 4.3).

Table 4.3 Spatial variation of throughfall and canopy characteristics for the funnel rain gauges. CPC = canopy projective cover; CAI = Canopy Area Index; BA = basal area.

Gauge	Cumulative throughfall (mm)	Cumulative deviation around site mean (mm)	Standard deviation (%)	CPC (%)	Mean CAI	BA (m^2/ha)	Distance to nearest stem (cm)	No. of trees*
1	365.5	-60	5.9	92.3	2.73	121.8	130	15
2	485	59.5	7.5	88.7	2.16	187.9	110	11
3	N/A	N/A	N/A	87.2	2.18	240.2	115	7
4	501.5	76	8.8	94.9	3.54	361	130	8
5	404	-21.5	5.7	93.7	7.05	47.3	40	4
6	N/A	N/A	N/A	90.2	2.5	64.4	70	13
7	438.5	13	6.9	89.5	2.24	219.5	90	6
8	478.5	53	7.3	87	2	55.9	50	13
9	322	-103.5	4.1	90.6	2.9	373	180	7
10	459	33.5	7.3	93.6	3.15	68	42	16
11	N/A	N/A	N/A	88.9	2.46	166.8	290	10
12	462	36.5	8.1	89.3	2.28	35.4	212	11
13	386.5	-39	6.6	90.9	2.35	56.7	90	18
14	374.5	-51	5.5	89.8	3.38	17.2	120	3
15	330.5	-95	5	95.7	4.72	16.3	7	7
16	567	141.5	7.8	90	2.55	12.3	95	9
17	427.5	2	6.9	91.5	2.92	240	120	15
18	476.5	51	8	93.2	2.86	85	160	12
19	160.5	-265	3.1	92.8	2.77	163.1	110	10
20	437.5	12	6.7	91.7	2.97	49	176	6
21	489.5	64	7.4	92.6	2.87	46.5	118	11
22	461.5	36	7.7	95.1	3.45	99.3	84	10
23	373.5	-52	4.9	96	4.01	15.7	99	8
24	392.5	-33	6.1	96.5	3.59	137.7	98	6
25	567.5	142	10.9	93	3.88	40.5	120	8
Mean	**425.5**	**0**	**6.74**	**91.8**	**3.1**	**116.8**	**114.2**	**9.8**

* Values taken for a 5 m radius around each gauge.

There were no significant relationships between the canopy characteristics and *Tf* distribution within the forest. There were also no significant relationships between the standardised *I* and CPC in the arc around the mean wind direction for each event. Therefore, the hemispherical canopy images taken at each rain gauge location (Figures 4.5 – 4.6) were examined visually to determine whether there were any obvious reasons why the distribution of rainfall around the site mean varied so greatly between the gauges.

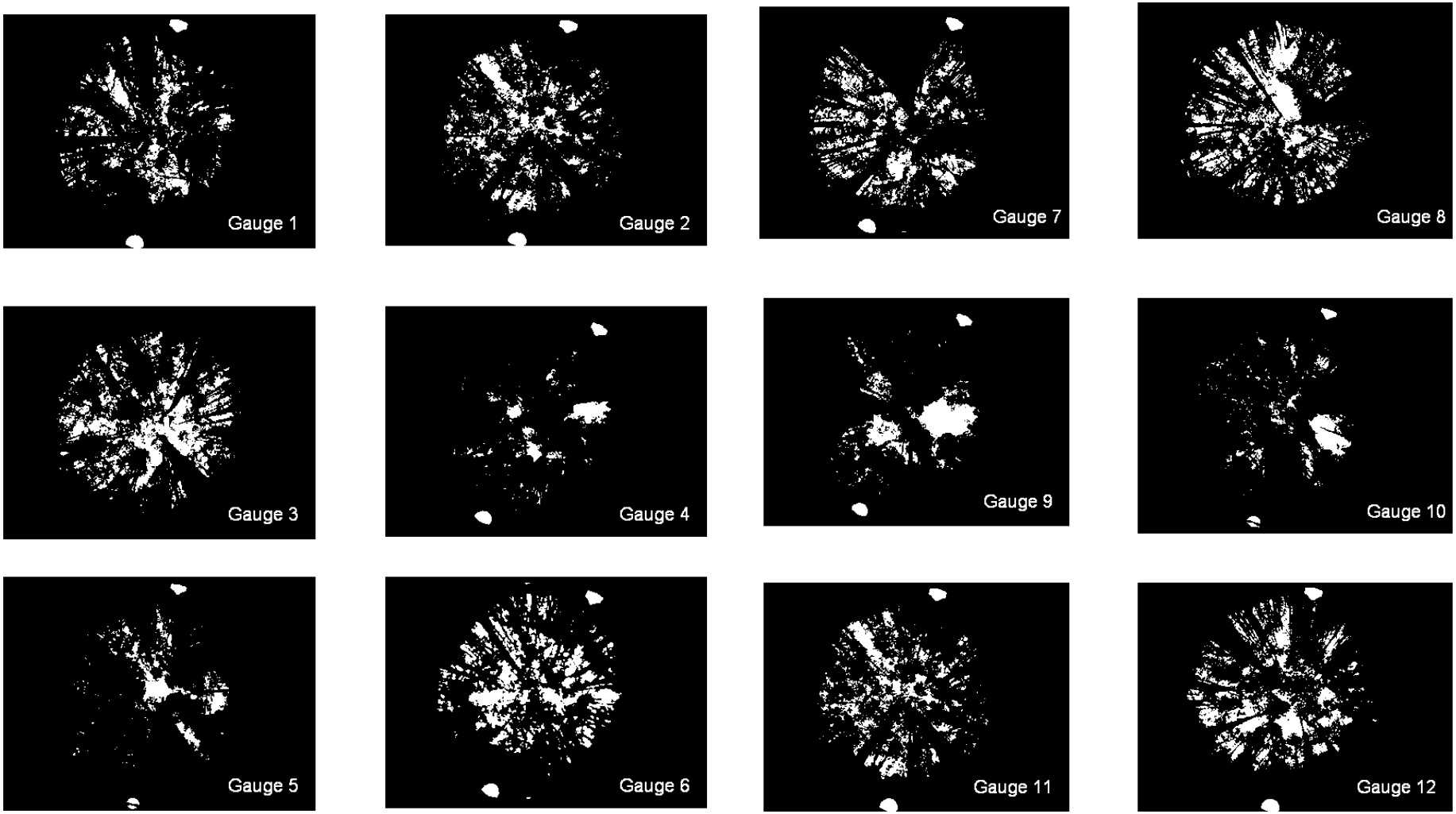

Figure 4.5 Hemispherical canopy images taken at the site of each funnel rain gauge for gauges 1-12. Top markers represent magnetic north, with east to the left of the image and west to the right of the image. Image date: 25/3/2011.

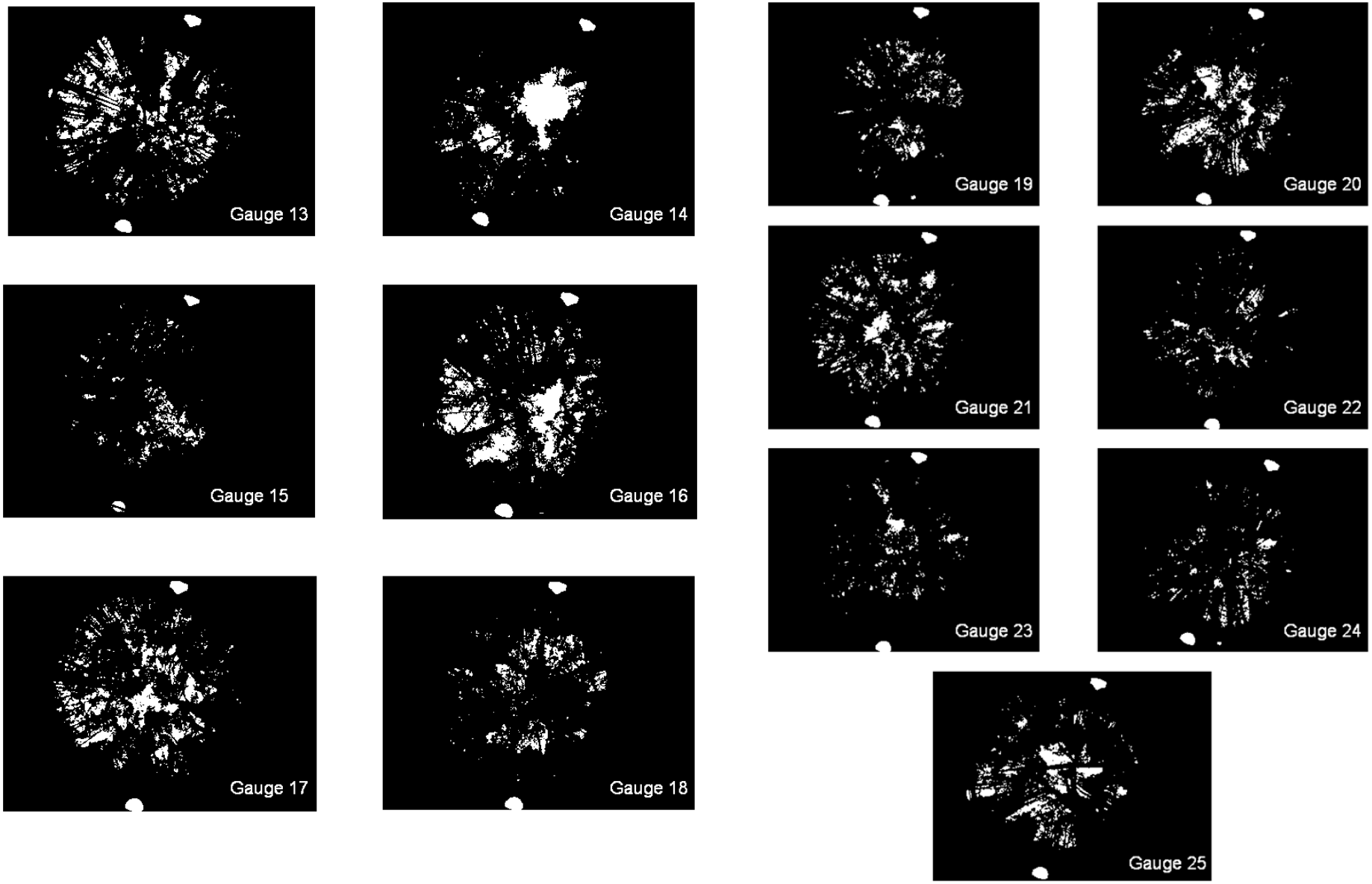

Figure 4.6 Hemispherical canopy images taken at the site of each funnel rain gauge for gauges 13-25. Top markers represent magnetic north with east to the left of the image and west to the right of the image. Image date: 25/3/2011.

It can be seen from the images that gauge 9, which recorded more than 100 mm below the site mean, has a lot of tree cover to the south-west, which would possibly block some of the rain from the prevailing weather direction. There is a large open section of the canopy towards the centre of the image, which would reduce the CPC, although not necessarily the rainfall (Figure 4.5). Gauge 16 recorded more than 140 mm of rain above the site mean. The image shows an open canopy, with small trees to the south-west that may be letting rainfall through (Figure 4.6). Gauge 17 was the gauge most similar to the site mean. The image shows a relatively open canopy in all directions, with a very even distribution of vegetation. There are also no large tree trunks in the image (Figure 4.6). The CPC and CAI values for gauge 17 are also very similar to the site means for these values (Table 4.3). Gauge 19 recorded the greatest deficit in rainfall around the site mean, at 265 mm below. The image shows a lot of tree trunks and thick cover, particularly within the southern quadrant of the photograph (Figure 4.6). Gauge 25 recorded more than 140 mm of rain above the site mean. The distribution of vegetation within the image looks reasonably even, and the CPC is close to the site mean. However, the image shows a low branch directly over the rain gauge, which could be responsible for funnelling rain through drip points directly into the gauge (Figure 4.6).

The distribution of species and basal diameter for all trees within a 5 m radius of each of the 25 rain gauges are shown in Figure 4.7. *Nothofagus cunninghamii* was evenly distributed around the site, with a number of larger sized trees. *Atherosperma moschatum* also has a strong presence, with a cluster of the species occurring exclusively around gauges 20, 24 and 25. In terms of basal area and number of individuals, *A. moschatum* is the dominant tree. *Eucryphia lucida* appears in clusters at about three of the mapped locations, while a handful of *Orites diversifolius* and *Anodopetalum biglandulosum* individuals occur within the mapped area of the site. Individual occurrences of *Cenarrhenes nitida* and *Phyllocladus aspleniifolius* also occur within the mapped area (Figure 4.7).

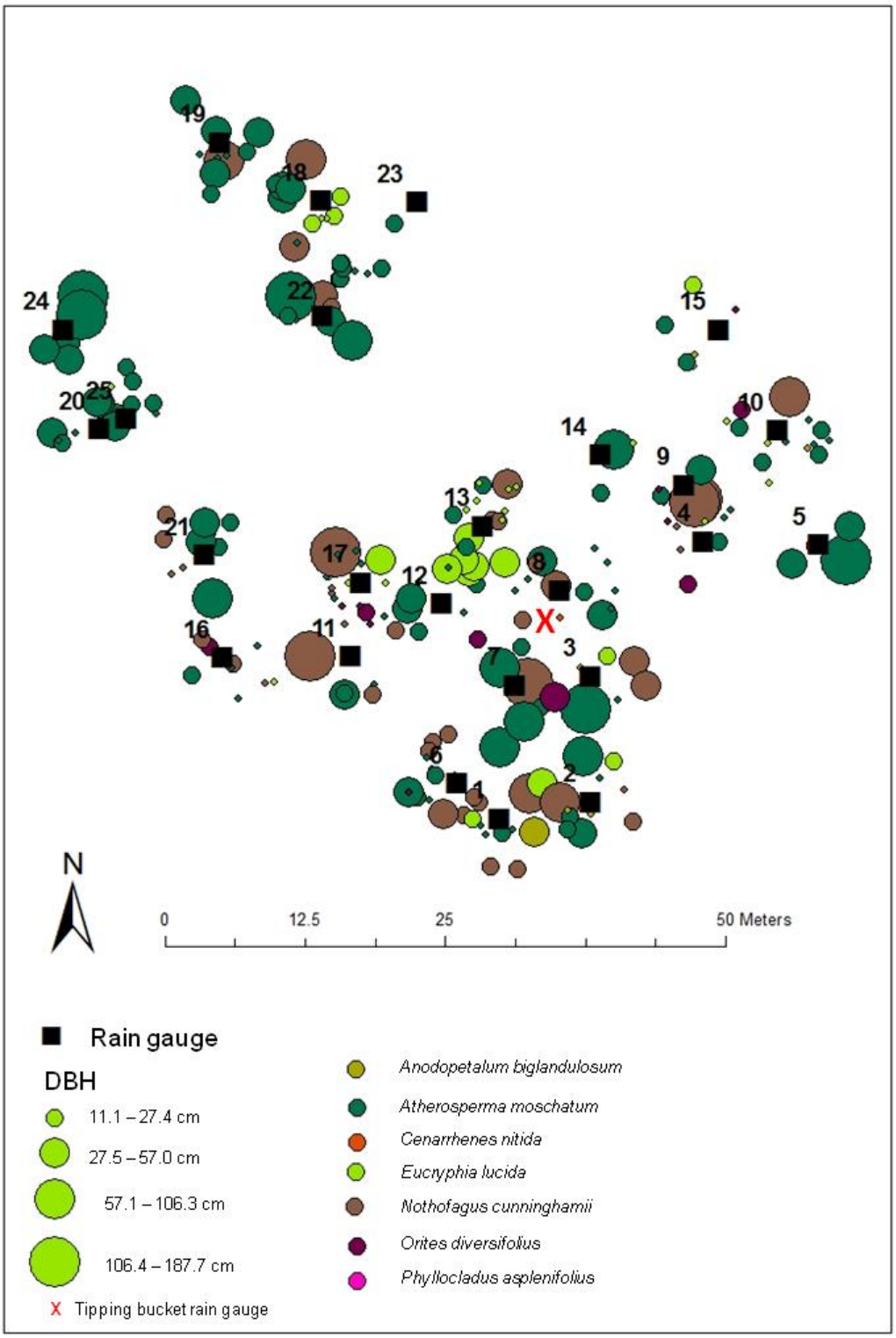

Figure 4.7 Map of the 25 rain gauges at Site Call with the location, species and diameter at breast height (DBH) in cm of all trees and tall shrubs in a 5 m radius of each rain gauge. Note: the apparent gaps in the above plan do not necessarily represent an absence of trees, rather they fall outside of the mapped area.

4.3.3 *Canopy saturation and interception*

The saturation value required before *Tf* occurs, as determined for the dry canopy events in Figure 4.4 is 1.48 mm at the tipping bucket rain gauge established within Site Call. When this value is adjusted by equation 4.4 the saturation value becomes 2.27 mm.

A *Pg* of 533 mm was collected during the study period whereas 420 mm of *Tf* was recorded within the rain gauge grid. Therefore, 21.1% of *Pg* was intercepted. The mean *I* recorded for all 11 events was 10.3 mm. This ranged from -9.4 mm for Event 10 and 37.8 mm for Event 2 (Table 4.2).

When the sub-events are included, *I* as a percentage of the *Pg* shows a high scatter for small events, with many of them having a negative *I* (Figure 4.8). Figure 4.8 also shows that for dry canopy events *I* as a percentage of *Pg* remains relatively constant regardless of *Pg*. The mean *I* as a percentage of *Pg* for dry events was 22.9%, ranging from 11.2% for Event 7 to 32.3% for Event 3.

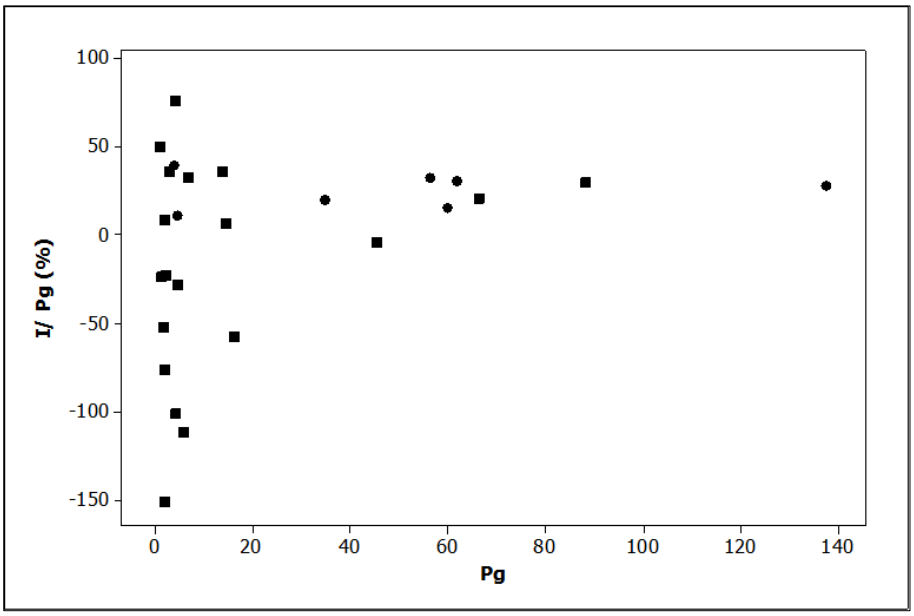

Figure 4.8 Relationship between interception (*I*) as a percentage of gross rainfall (*Pg*) and gross rainfall (*Pg*) for all events and sub-events. Squares = wet events; circles = dry events.

The relationship between *Pg* and *Tf* for all events and sub-events was very strong (*R*-sq = 0.94; *P*-value <0.001; Figure 4.10). All dry canopy events fell either on or below the regression line

with the exception of Event 6. Small wet canopy events were distributed both above and below the line, whereas the larger wet canopy events mostly occurred above the line (Figure 4.9).

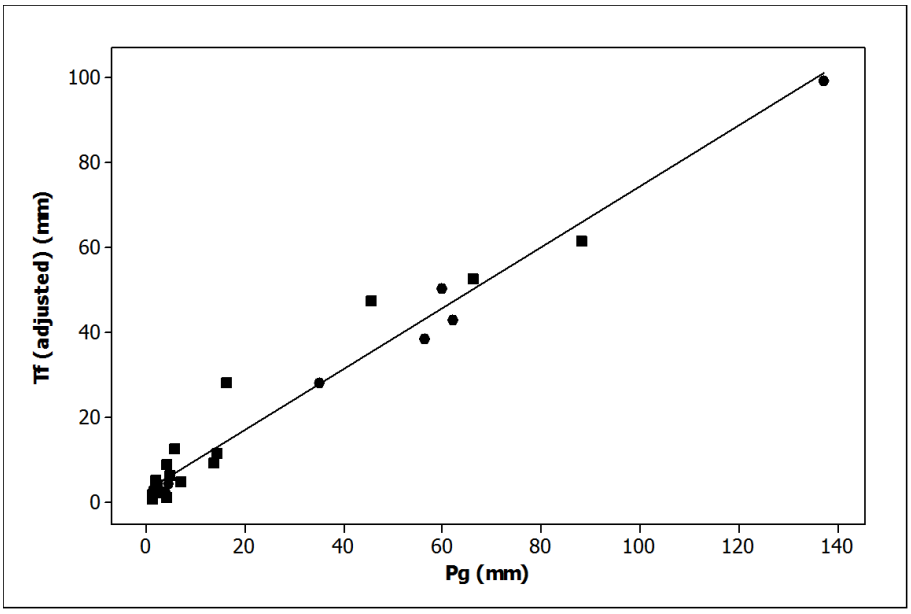

Figure 4.9 Gross rainfall (*Pg*) and throughfall (*Tf*) for all events and sub-events. Squares = wet events; circles = dry events.

There is a strong relationship between *I* and *Pg* (Figure 4.10) with the polynomial regression fit providing the strongest fit with the data (*R*-sq = 0.87; *P*-value <0.001). All the dry canopy events except Event 6 fall above the regression line, with many of the wet canopy events falling below the line. Many of the smaller events show a negative interception (Figure 4.10).

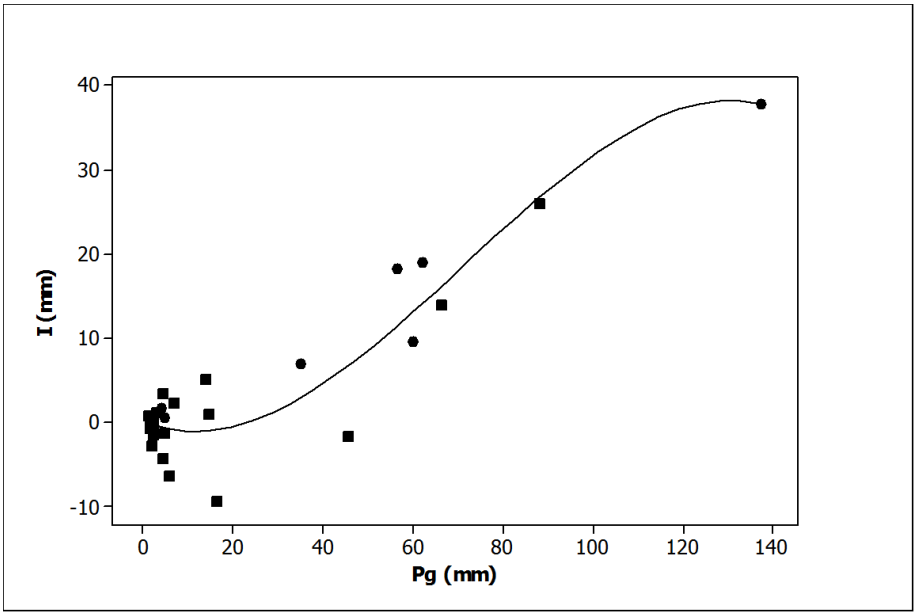

Figure 4.10 Gross rainfall (*Pg*) and interception (*I*) for all events and sub-events. Squares = wet events; circles = dry events.

There was a positive linear relationship between *I* as a percentage of *Pg* and the rainfall rate (*R*-sq = 0.35; *P*-value = 0.002) when the outlier for sub-event 8.3 was excluded from analysis (Figure 4.11). The relationship between *I* as a percentage of *Pg* to rainfall rate remained relatively constant regardless of rainfall rate. All the dry canopy events except Event 6 fell above the regression line (Figure 4.11).

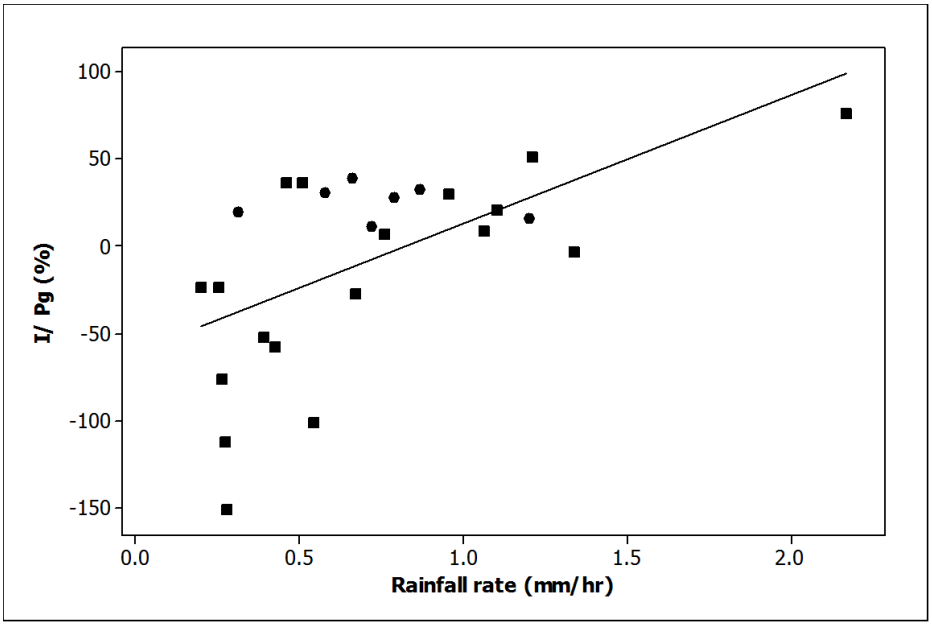

Figure 4.11 Relationship between interception (*I*) as a percentage of gross rainfall (*Pg*) and rainfall rate. Squares = wet events; circles = dry events.

4.4 Discussion

The number of funnel rain gauges required to adequately sample a study site while keeping the mean relative error low appears to vary widely among forest types. Puckett (1991) found that the mean relative error could be kept below 10% in mixed hardwood forests with as few as 11 gauges, with 22-23 gauges resulting in a mean relative error of less than 5%. Carlyle-Moses *et al.* (2004) found that the mean relative error could be kept below 5% with 9 gauges in a red oak forest in Mexico. Zimmermann *et al.* (2010), on the other hand, found that even with 200 gauges the error in sampling exceeded 10% for the majority of events. The coefficient of variation tends to decrease with increasing rainfall (Leyton *et al.* 1967; Staelens *et al.* 2006; Zimmermann *et al.* 2010). Therefore, the number of collectors required to increase the accuracy of capture is related to the size of the rainfall events. As the present study mostly consisted of longer term capture resulting in large rain events 23-25 collectors was thought to be sufficient. As a consequence, the smaller sub-events should be viewed with caution. This is apparent in the wide scatter of results for events of less than 10 mm of rainfall (Figure 4.8).

Although efforts were made to define each rain event by a 24 hour gap between showers this did not always occur. Twenty-four hours was the time period used as it was thought that this would be an adequate period of time for the canopy to become fully dry. Although this might be true during certain summers, this assumption proved false during the second collection period in autumn, when day lengths were shortening, and the weather was overcast for much of the time. The 24 hour time period was also used in keeping with the Soil Dryness Index (SDI), and other similar indices, which are calculated on a 24 hour basis.

One fifth of the total *Pg* that was collected during the study period was intercepted. This is a similar value to those found in other studies of interception for a broad range of vegetation types (for example; 31.7% for a *Pinus sylvestris* plantation in south-east England (Rutter 1963); 35.3% for a *P. nigra* plantation in south-east England (Rutter *et al.* 1972); 27.2% for semi-arid shrubland in Mexico (Návar and Bryan 1990); 24-38% for *P. radiata* plantation in New South Wales (Pook *et al.* 1991); 32% for Mitchell grassland in western New South Wales (Dunkerley and Booth 1999); 22.8-25% in Douglas-fir–Western Hemlock old-growth forest in the Pacific Northwest USA (Link *et al.* 2004); 21% for mixed deciduous forest in Belgium (Staelens *et al.* 2006)). The canopy interception took the form of water held within the canopy, water evaporated from the canopy and water held and possibly transported along the stems of trees. The collection of data from two events and 8 sub-events, which recorded negative values for *I*, suggests that there may

have also been a spatial variability in the distribution of rain between the control and study site, and that this is likely to have affected the derivation of *I*.

As the collection of rainfall data occurred across spring/summer 2010/11 and autumn 2012 it was assumed that a wide range of evaporative conditions would have been captured and that the data are therefore representative of variation in *E*. The absence of winter rainfall data is not considered a limitation as *E* during winter would be minimal and winter is not a period of interest from a fire management perspective. Furthermore, Klaassen *et al.* (1998) found that, on average, evaporation during rainfall was only 4.6% of precipitation in a Douglas-fir forest in the Netherlands and that water storage was the dominant process in the interception of rainfall. Therefore, the contribution of *E* to *I* is likely to result in a variation of less than 5%.

S_f and water held on the stems of trees may be contributing to some of the interception value. S_f was not measured in this study due to the very small contribution of S_f in other structurally similar forests (Crockford and Richardson 2000) as well as the fact that it was not thought that the morphology of the trees was conducive to S_f (Crockford and Richardson 1990). Furthermore, the great mass of mosses and lichens found on the trunks and branches of callidendrous rainforests is likely to be preventing any water running down the trunks from reaching the soil profile. Link *et al.* (2004) found that mosses and lichens can increase approximately 6-10 times their dry weight. While Dunkerley (2000) points out the important role of S_f in dry regions as a way of transporting water to the root zone to increase the efficiency of light rainfall, in wet forests it is possible stems have not adapted to provide efficient S_f as water is not a limiting factor.

Negative values of *I* are not uncommon and have been reported from a wide range of forest types (Crockford and Richardson 2000). There are a number of explanations that may account for negative interception results. Perhaps the most likely explanation in this study is the occurrence of small, isolated showers that fell at Site Call but were not recorded at the control site, three kilometres away. Crockford and Richardson (2000) state that although many researchers acknowledge the difficulty in measuring *Tf*, problems in measuring *Pg* are commonly ignored and that one such problem is the ideal positioning of the rain gauge. Although attempts were made to address this issue with the adjustment of data from the control gauge using a gauge close to the study site, errors in the capturing of small localised events were not mitigated. With the exception of Events 1 and 10, all the occurrences of negative *I* were from sub-events, and therefore events with low levels of rainfall. This supports the theory of small and localised

showers. Link *et al.* (2004) found that spatial variation in *I* was observed to increase steeply for small events (<10 mm). The present study shows a wide variation in *I* as a percentage of *Pg* for events less than 10 mm (Figure 4.9). All the sub-events that recorded negative *I* were less than 10 mm in size. Of the larger events, Event 1 was 45 mm, however the variation between the *Pg* and *Tf* was less than two mm, or 3.7%. Only Event 10 recorded greater than 10 mm of *Pg* and had a considerable variation between *Pg* and *Tf*, of more than 9 mm, or 58%.

Another explanation that may account for negative *I* is drifting fog, mist or cloud, which could be enhancing *Tf*. This explanation was used to explain negative *I* values obtained by Pook *et al.* (1991) in a *Eucalyptus viminalis* forest in New South Wales. Carlyle-Moses *et al.* (2004) also found that cloud-combing was increasing *Tf* in a Mexican red oak forest, without increasing *Pg*. In this study the events that recorded negative *I* were all events where the canopy was wet at the start of the event. Consequently the canopy was already fully or partially saturated and much of the additional rain would contribute to *Tf*. This may partially explain a higher value in *Tf*. Wind shake may also be contributing to higher *Tf*, reducing the amount of water that can be held in the canopy (Jackson 1975). However, wind shake is unlikely to contribute an extra 9 mm, or 58%, to the *Tf* recorded for Event 10. Furthermore, wind shake occurs during high winds (Jackson 1975) and none of the recorded wind speeds during the study period were particularly high, with Event 10 being mid-range.

S was determined from the amount of *Pg* that fell before *Tf* was registered for 5 dry canopy events. This is similar to methods used by Rutter (1963). A number of studies (e.g. Jackson 1975; Link *et al.* 2004) have used the Leyton method, which subjectively identifies the inflection point in the *Tf* – *Pg* relationship and a line is fitted to events above the inflection point. *S* is then determined from the intercept of this line on the x-axis (Leyton *et al.* 1967). However, due either to the small number of total events or the lack of a sufficient number of small rainfall events, no inflection point was apparent in the present study. Therefore, *S* was determined from field estimates. The number of events that could be used in this process was hindered by the resolution of the rain capture data, which had to be offset by the capacity of the data loggers to store a sufficient quantity of data between visits. However, the quality of the values used is thought to be good, as once *Tf* commenced it was continuous, indicating that these values represent genuine *Tf* rather than just the T_f component. The value for *S* recorded for this study was 2.27 mm. This is within the range of results from other studies (Rutter 1963; Bell and Gatenby 1969; Rutter *et al.* 1975; Herwitz 1985; Pook *et al.* 1991; Klaassen *et al.*; 1998; Link *et al.* 2004). Henceforth in this study, estimates of *S* will be rounded to two millimetres in order to

avoid any false suggestions of high precision. This is further justified as it is likely that values for *S* will be affected by additional variables such as temperature, which affects the viscosity of the water; and wind (Jackson 1975). Furthermore, although *S* is considered to be a constant term in evergreen canopies (Rutter *et al.* 1975; Gash *et al.* 1980), Link *et al.* (2004) found variations between seasons due to phenological changes in the LAI as well as storm damage to the forest. Crockford and Richardson (1990) found that values for *S* for the same species varied significantly between years. They attributed this to drought stress, with such trees having older leaves that were less hydrophobic than trees not suffering from drought. From a management perspective this suggests that at least two millimetres of rain should be allowed to fall before derivations of the SDI, or similar indices, within a rainforest are undertaken, and that approximately 20% of the rainfall from events above two millimetres will be intercepted.

The expected lag between *Pg* and *Tf* was observed in Figure 4.5. The diagrams show the canopy intercepting small amounts of rainfall, while larger quantities are seen passing through the canopy after a short lag period. In almost all cases *I* is apparent as the *Tf* recorded is less than *Pg*. However, for the event between 6 and 11 April 2012 there is a spike in *Tf* that is not matched by *Pg*. This is most likely due to the spatial variation previously discussed, with an isolated shower falling at the study site that was not registered at the control site.

Event 6 appears to stand out in much of the analysis. Figure 4.11 shows Event 6 recording less *I* for the amount of *Pg* than is expected, based on the other dry canopy events. Figure 4.12 shows Event 6 recording a smaller *I* percentage for its rainfall rate than is expected, based on the other dry canopy events. Event 6 consisted of 60 mm of rain that fell between 17 and 22 February 2011. Along with Event 5 it is the only dry canopy event to be intermittent (i.e. has a gap of at least 24 hours without rainfall during the event). Event 6 recorded the highest rainfall rate for the dry events and the highest average wind speed and maximum wind gust for all events (Table 4.2). Nine mm of the total rain that fell during Event 6 was intercepted, or 15.8%, which is below the average *I* for the whole study period. The reasons why Event 6 recorded less *I* than would be expected during dry canopy events for both *Pg* and rainfall rate, are not entirely clear. It could be due to wind shake as Event 6 recorded the highest maximum wind gust, however, other similarly high wind gusts were recorded for Events 3, 4 and 5, which were also dry canopy events. Also, the lack of any significant relationships between any of the wind variables and *I*, suggests that this might not be so.

Fifty-nine percent of the rain gauges recorded *Tf* that was equal to, or above, the site mean, while 25% of gauges recorded *Tf* equalling or exceeding *Pg*. This is undoubtedly partly due to gaps or drip points in the canopy cover (Hutjes *et al.* 1990; Bellot and Escarre 1998; Link *et al.* 2004; Zimmermann *et al.* 2010). The lack of a significant result between CPC and *Tf* in this study is possibly due to the high values of CPC, which are not particularly variable among the gauges, ranging from 87% at gauge 8 to 96.5% at gauge 24. Staelens *et al.* (2006) found a significant negative correlation between CPC and *Tf*. Although their mean CPC was greater than for this study (94% rather than 91.8%) their range was also greater (81 – 99.7%). It is interesting that the average CPC in this study is high (91.8%) whereas the CAI is relatively low (3.1) compared with other studies, and much lower than would be expected for a closed canopy temperate rainforest. This is likely a feature of the forest structure, as the vertical profile of a callidendrous rainforest is relatively simple compared to other dense forests. It is also possible that the seemingly low value for CAI is due to different methods used to derive the values, as none of the other CAIs cited in this study were derived from hemispherical canopy images.

The basal area, the distance to the nearest stem and the number of individuals in a 5 m radius around each rain gauge also showed no relationship to the variability of *Tf*. This is less surprising as it is assumed that these variables would have less of a relationship to *Tf* than do the direct measures of canopy cover. However, in a structurally simple plantation of Sitka Spruce, Ford and Deans (1978) found a relationship between *Tf* and proximity to the stem, with the largest rates of *Tf* occurring in areas close to the tree stems. They related this to the lack of overlapping branches from other trees. Conversely, Johnson (1990) found the greatest *Tf* occurring at the edges of the tree canopies in a 50-year-old Sitka spruce forest that had been trimmed. The lack of a relationship between *Tf* and proximity to the tree stem is not so surprising in an unmodified, structurally complex forest, and similar results have been found in other natural forests (Carlyle-Moses *et al.* 2004). The same authors also failed to find a relationship between the basal area and *Tf*.

It is possible that the direction and angle of rain, which are determined by wind direction and speed, are more important in explaining *Tf* variation than any of the canopy variables. However, no relationships were found between the wind variables and *Tf* or for *Tf* and the CPC from the arc around the mean wind direction for each event. The absence of a significant result in this case may once again be due to the small range of wind directions observed. The dominant wind direction in south-west Tasmania is south or west, with northerly winds occurring occasionally during the summer and easterly winds being a rarity. During this study, only Event 1 recorded a

mean wind direction from the north-west with the mean wind directions for all other events being from the south-west. In most studies examining *S* wind has been excluded (Dunkerley 2000), and no studies could be found that examined wind as a factor in determining the spatial distribution of *Tf*, although it is likely to play an important role. Furthermore, although there was little variation in the range of canopy cover values, Figure 4.8 shows that the forest occurring within a 5 m radius of each rain gauge is not homogenous, with sectors dominated by either *A. moschatum*, *E. lucida*, or a mixture of *A. moschatum* and *N. cunninghamii*. Although no quantitative measures explained the spatial distribution of *Tf* a visual examination of the hemispherical canopy images identified possible drip points as well as other hindrances that might explain the *Tf* distribution, especially with reference to the prevailing wind direction.

The results highlight issues that may arise in rainforest management during fire periods. As a result of the variable distribution of *Tf* some patches of the rainforest floor are receiving considerably less water than others, which will result in patchy fuel dryness. Patterns in *Tf* affect the heterogeneity of hydrological processes on the forest floor (Staelens *et al.* 2006). If these patterns in *Tf* remain constant over time, soil moisture will be higher than average in some places, while other patches would be considered as dry soil sites in the same forest (Bellot and Escarre 1998). Therefore, if spotting occurs during bush fire conditions, it is possible that a spot will result in fire ignition within one of the dry patches of the rainforest floor. Even if this fire is of low intensity considerable damage may occur as fires that trickle through the understorey of rainforest have been observed to result in the later slow death of the canopy species. Although this fire would be unlikely to spread very far, assuming the surrounding patches are of a sufficient fuel moisture to extinguish the fire, these spot fires could create a positive feedback whereby the integrity of the rainforest is reduced, opening the canopy, in turn creating larger areas of low fuel moisture, which increases the flammability of the rainforest into the future.

The study site was thought to be broadly representative of cool-temperate callidendrous rainforest. As temperate rainforests contain a very small number of tree species, and there is great conformity within callidendrous rainforest, it was thought that the results obtained could be generally extrapolated to other areas of callidendrous rainforest in Tasmania than if an alternative, more variable form of rainforest had been chosen. Both Jackson (1975) and Herwitz (1985) point out the considerable issues with extrapolating results of interception studies to other forests. In the case of Herwitz (1985) this was due to the significant variation in the water holding capacity of canopy tree species in a tropical rainforest. As Herwitz (1985) mentions, tropical rainforests typically exceed 100 tree species per hectare, whereas temperate rainforests

rarely contain more than 5 tree species per hectare. Therefore, the small number of canopy species and the relative homogeneity of the forest canopy are likely to make the results of the present study more applicable to other cool-temperate rainforests within Tasmania, than results from one site in tropical forest to tropical forest as a whole.

Chapter 5 – Predicting rainforest microclimate

5.1 Introduction

Oke (1978) defines a microclimate as a component of the atmosphere that occurs on a scale of 0.01 to 1000 m. However, as the atmosphere does not occur in discrete parcels, definitions of the dimensions of microclimate are somewhat arbitrary (Oke 1978). Stoutjesdijk and Barkman (1992) define the microclimate as only occurring in the lower two metres of the atmosphere and the upper one metre of the soil profile, whereas other definitions of microclimate have a greater vertical extent (Geiger 1965; Oke 1978; Read 2005). For example, Oke (1978) considers that a forest canopy can act as a barrier between components of the atmosphere, with microclimate pertaining to the air beneath this barrier.

The microclimate of a forest can be considered to consist of three components; the canopy area, the trunk area, and the forest floor (Geiger 1965). Temperature variations are much greater within the canopy than they are in the trunk area, although nowhere near as variable as temperatures in the open (Geiger 1965). Wind penetrates the canopy to a greater degree than the trunk and forest floor areas, resulting in instability compared to the forest floor and trunk zones (Geiger 1965). The present study is concerned with the trunk area, forest floor and top 30 cm of the soil profile as this area is the domain of the forest litter and undergrowth, two important components of the forest that affect the likelihood of fire. The microclimatic variables relevant to fire hazard are sunlight exposure, wind exposure, precipitation, temperature (of air and soil) and moisture content (of air and soil; Davies-Colley *et al.* 2000).

5.1.1 Rainforest microclimate

The atmosphere in the interior of a rainforest is more humid and cool during the day than the atmosphere above the rainforest canopy (Geiger 1965). Light levels and wind speeds are also less within the rainforest (Davies-Colley *et al.* 2000; Read 2005; Pohlman *et al.* 2008; Ray *et al.* 2010), due to its dense canopy (Adam 1992; Bowman 2000; Read 2005). As a result of this cool, humid microclimate, fire is uncommon (Barker 1991; Bowman 2000; Ray *et al.* 2010), with sustaining fire within rainforest being rarer still (Ray *et al.* 2010). However, models predict future drier

conditions in summer for western and south-western Tasmania (Grose *et al.* 2010), making it important to understand the conditions in which fire will sustain in rainforest. Once these conditions are understood, fire managers will need to be able to deduce their incidence from data from meteorological stations. Thus, this chapter is devoted to predicting the elements of the rainforest microclimate relevant to flammability from external weather data.

Meteorological data have been used successfully to interpolate weather conditions both spatially and temporally (Parishwad *et al.* 1998; Bhakar *et al.* 2008; von Platen *et al.* 2011; Yasar *et al.* 2012). In many of these studies there is often substantial distances between the regions, which is overcome by incorporating geographical variables such as latitude, longitude and altitude into the models (Yasar *et al.* 2012). Other studies have examined differences between microclimates (Chen *et al.* 1993; Chen *et al.* 1995; de Freitas and Enright 1995; Davies-Colley *et al.* 2000) but have not attempted to use these data sets in a predictive capacity.

5.2 Methods

The climate data described in Chapter 2 were used for data analysis.

To ensure that the different sensors were accurate, regression analysis was used to compare the temperature and humidity data from the ibutton temperature loggers and the automatic weather station at Site Call for the period between 21 September 2010 and 6 December 2010.

Saturation vapour pressure was estimated from air temperature using the following formula:

$$e_s = 0.597 + (0.05318 * T) + (0.000443 * T^2) + (0.000061 * T^3)$$

Equation 5.1

where e_s is the saturation vapour pressure and T is the air temperature.

The actual vapour pressure was then estimated from the following formula:

$$e = (RH * e_s)/100$$

Equation 5.2

where e is the actual vapour pressure and RH is the relative humidity expressed as a percentage.

Vapour pressure deficit (VPD) was determined by subtracting the actual vapour pressure from the saturation vapour pressure.

The air temperature at all sites was adjusted for altitude using the global mean environmental lapse rate of 6.5 °C/km (Wolfe 1964).

For each of the climatic variables for each measurement at each site, the rainforest value was subtracted from the control value (Δ). The Δ for air temperature and VPD during fire weather conditions (temperatures > 25 °C; relative humidity < 25%) were also determined. This procedure could only be done for Site Call, as it was the only site with a sufficient amount of such data. Similarly, the Δ for air temperature and VPD at all sites during sunny conditions (total solar radiation > 1.5MJ/M^2 in each hour) were determined. As the dominant tree at Site DB is deciduous the Δ for all variables was also determined for the summer months only.

The Δ for all of the climatic variables from each site were correlated with the corresponding variable from the control using Pearson's product moment correlation coefficient.

5.3 Results

The level of agreement between the ibutton loggers and the temperature data from the automatic weather station at Site Call was very good (*R*-sq = 0.99; *P*-value <0.001).

$$ibuttonT_{air} = 0.4216 + (0.9511 * AWST_{air})$$

Equation 5.3

where $ibuttonT_{air}$ is the air temperature recorded by the ibutton loggers and $AWST_{air}$ is the air temperature recorded by the automatic weather station.

The relative humidity relationship was not as strong but still highly significant (*R*-sq = 0.88; *P*-value <0.001).

$$ibuttonRH = 162.3 - (5.837 * AWSRH) + (0.1008 * AWSRH^2) - (0.000487 * AWSRH^3)$$

Equation 5.4

where *ibuttonRH* is the relative humidity recorded by the ibutton loggers and *AWSRH* is the relative humidity recorded by the automatic weather station.

When air temperature was adjusted by the global mean environmental lapse rate, the temperatures at Site Call were adjusted downwards by 0.58 °C, temperatures at Site Imp were adjusted downwards by 0.49 °C and temperatures at Site DB were adjusted downwards by 0.07 °C.

The correlation analysis of Δ with the corresponding variable from the control showed that as each control variable had higher values there was a decline in Δ (Table 5.1; Figure 5.1). The relationship between Δ temperature at Site DB and the control temperature was significant (Table 5.1), although it was less steep than for the other sites (Figure 5.1 *c*). This was also the case for VPD at Site DB (Figure 5.1 *f*). As the temperature at the control increased the Δ ranged from warmer within the forest to warmer at the control (Figure 5.1 *a* & *b*). Similarly, as VPD increased at the control the Δ VPD ranged from fully saturated air at both sites to a greater VPD at the control. The Δ wind speed ranged from almost identical wind speeds between the control and forest when wind speed was zero to much faster wind speeds at the control as the wind speed increased (Table 5.1; Figure 5.1).

Table 5.1 Pearson's product moment correlation coefficients between Δ of each variable inside and outside the forests and the variable at the control. Significant relationships are shown in bold.

Variable	Pearson's product moment correlation coefficient
Δ air temperature Site Call	**-0.697**
Δ air temperature Site Imp	**-0.818**
Δ air temperature Site DB	**-0.127**
Δ vapour pressure deficit Site Call	**-0.934**
Δ vapour pressure deficit Site Imp	**-0.955**
Δ vapour pressure deficit Site DB	**-0.263**
Δ wind speed Site Call	**-0.941**
Δ wind speed Site DB	**-0.949**

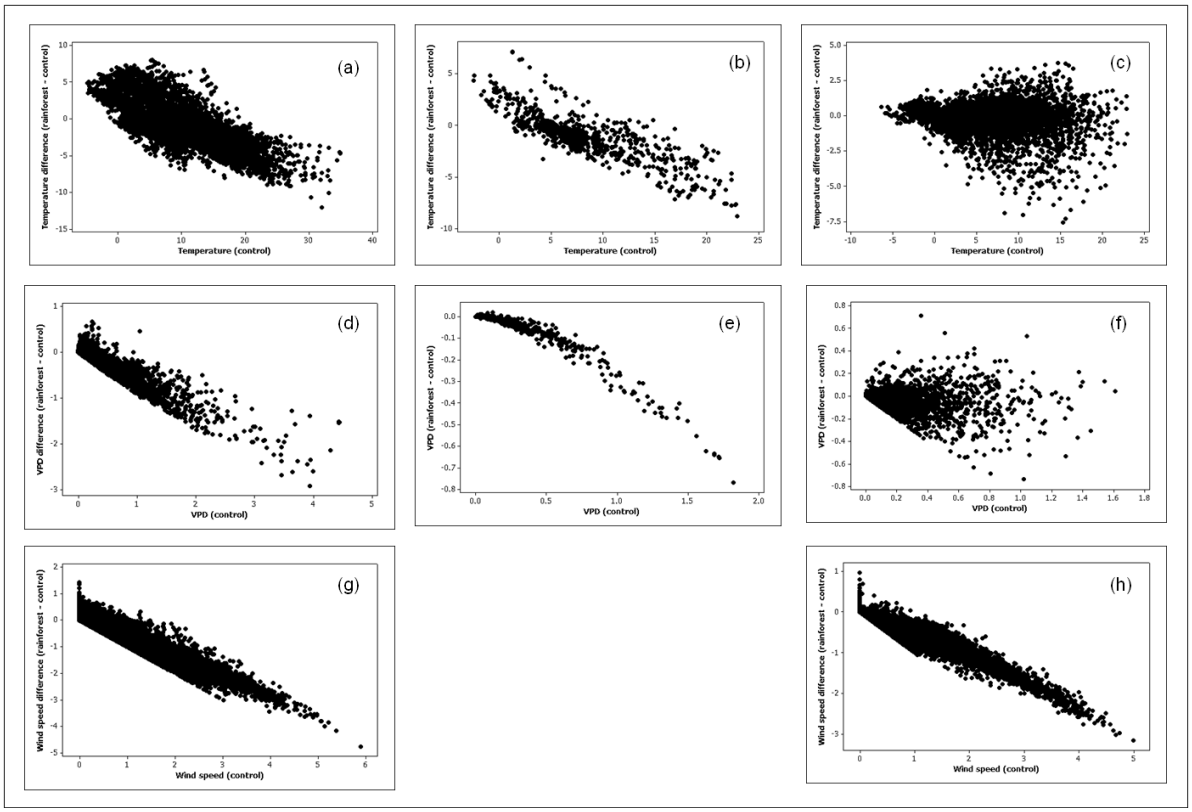

Figure 5.1 Relationship between Δ for each variable and the corresponding variable at the control: (a) air temperature (°C), Site Call; (b) air temperature (°C), Site Imp; (c) air temperature (°C), Site DB; (d) vapour pressure deficit (VPD; kPa), Site Call; (e) vapour pressure deficit (kPa), Site Imp; (f) vapour pressure deficit (kPa), Site DB; (g) wind speed (m/s), Site Call; (h) wind speed (m/s), Site DB.

5.3.1 Air temperature

When temperatures were greater than 25 °C and relative humidities less than 25% at the control for Site Call the median Δ temperature was 6.84 °C cooler within the forest while the mean was 6.79 °C cooler, with a range in Δ from -13 to -3 °C showing that the forest was always cooler than the control (Figure 5.2). For all temperatures, the median Δ temperature was 1.7 °C cooler within the forest while the mean was 1.6 °C cooler, with Δ ranging from highly positive to highly negative, indicating that sometimes the forest was cooler than the control (Figure 5.3). The median Δ temperature for measurements when total solar radiation was greater than 1.5 MJ/m2 at the control was 3.9 °C cooler within the forest, while the mean was 4.4 °C cooler, with a range in Δ from -9 to -1.5 °C, indicating that under these conditions the forest was always cooler than the control(Figure 5.4).

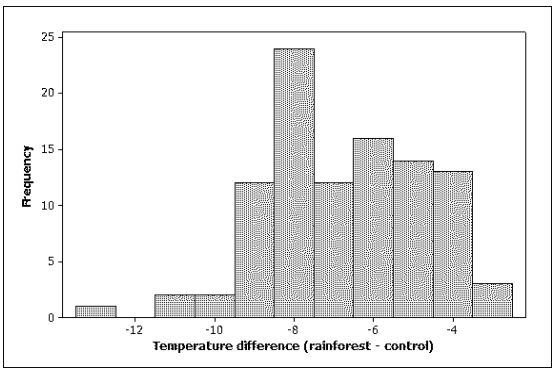

Figure 5.2 Frequency histogram of the Δ between inside and outside forest conditions for all temperatures greater than 25 °C and relative humidities less than 25% at Control 1, for Site Call and Control 1.

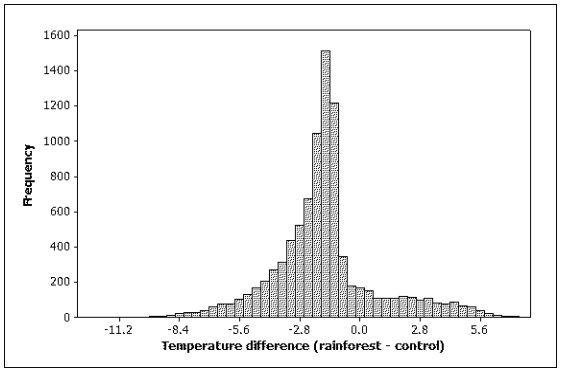

Figure 5.3 Frequency histogram of the Δ between inside and outside forest conditions for all temperatures (°C) at Site Call and Control 1.

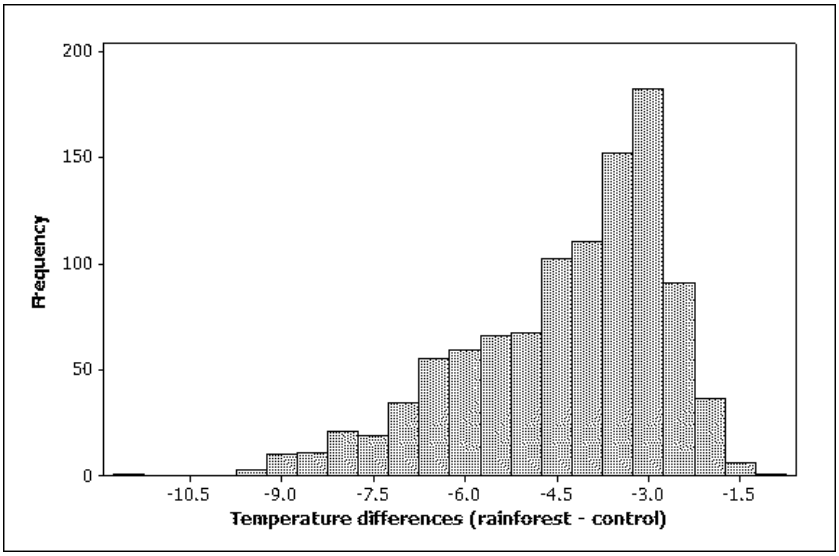

Figure 5.4 Frequency histogram of the Δ between inside and outside forest conditions for all temperatures (°C) at Site Call and Control 1 when the total solar radiation is greater than 1.5 MJ/m² at Control 1.

For all temperatures at the control for Site Imp, the median Δ temperature was 1.3 °C cooler within the forest while the mean was 1.5 °C cooler, with a range from -9.3 to 6.7 °C (Figure 5.5). The median Δ temperature for measurements when total solar radiation was greater than 1.5 MJ/m^2 at the control was 3.9 °C cooler within the forest, while the mean was 4.3 °C cooler, with a range in Δ from -9.3 to -1.2 °C (Figure 5.6).

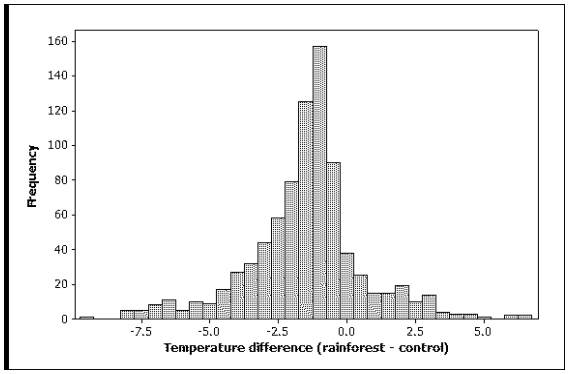

Figure 5.5 Frequency histogram of the Δ between inside and outside forest conditions for all temperatures (°C) at Site Imp and Control 1.

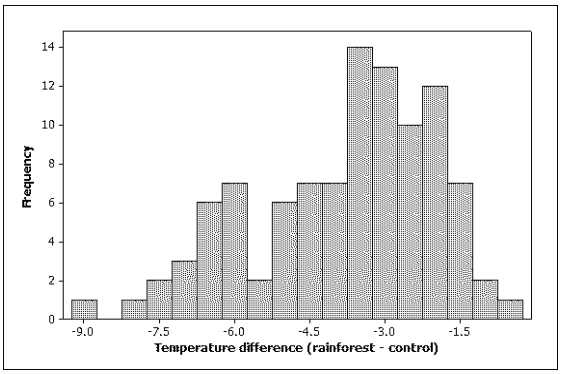

Figure 5.6 Frequency histogram of the Δ between inside and outside forest conditions for all temperatures (°C) at Site Imp and Control 1 when the total solar radiation is greater than 1.5 MJ/m² at Control 1.

For all temperatures at the control for Site DB, the median Δ temperature was 0.17 °C cooler within the forest while the mean was 0.29 °C cooler, with a range from -7.6 to 3.7 (Figure 5.7). The median Δ temperature for measurements when total solar radiation was greater than 1.5 MJ/m^2 at the control was 0.93 °C cooler within the forest, while the mean was 0.99 °C cooler with a range in Δ from -7.4 to 3.6 °C (Figure 5.8). When only the summer months were included in analysis the median Δ temperature was 0.16 °C cooler within the forest, while the mean was 0.25 °C cooler, with a range from -7.6 to 3.7 °C (Figure 5.9).

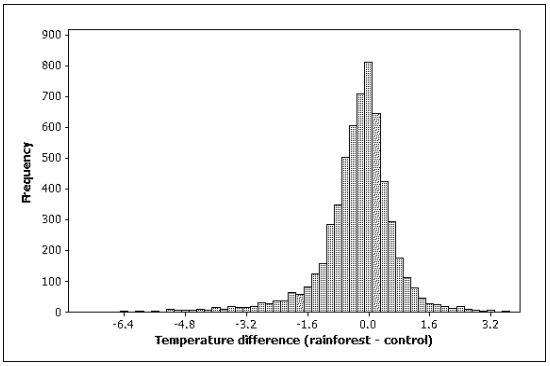

Figure 5.7 Frequency histogram of the Δ between inside and outside forest conditions for all temperatures (°C) at Site DB and Control 2.

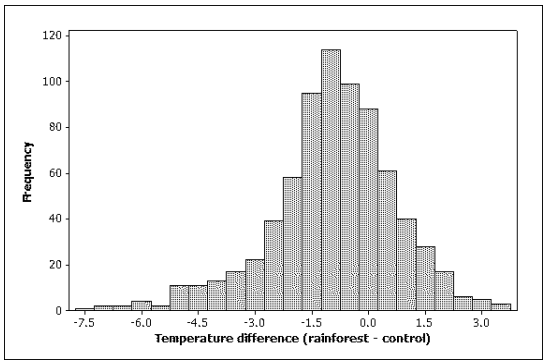

Figure 5.8 Frequency histogram of the Δ between inside and outside forest conditions for all temperatures (°C) at Site DB and Control 2 when the total solar radiation is greater than 1.5 MJ/m² at Control 2.

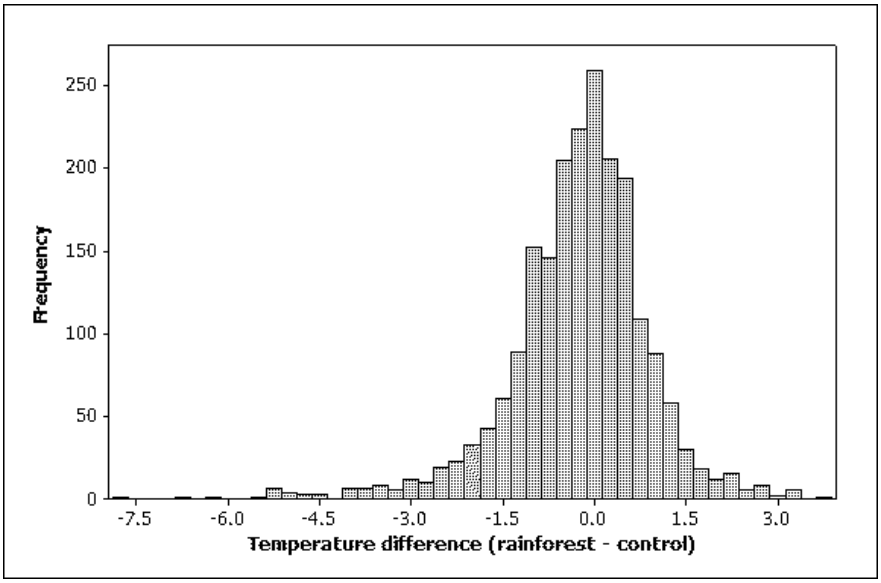

Figure 5.9 Frequency histogram of the Δ between inside and outside forest conditions for all temperatures (°C) from 1/12/2010 to 28/2/2011 for Site DB and Control 2.

5.3.2 Vapour pressure deficit

When temperatures were greater than 25 °C and relative humidities less than 25% at the control for Site Call the median Δ VPD was 1.95 kPa while the mean was 1.97 kPa, with the range in Δ VPD from 0.83 to 2.91 kPa, showing that the forest was always more humid than the control (Figure 5.10). For all values of VPD at the control, the median Δ VPD was 0.08 kPa while the mean was 0.16 kPa, with Δ VPD ranging from -2.9 to 0.7 kPa, indicating that sometimes the forest was more humid than the control (Figure 5.11). The median Δ VPD when total solar radiation was greater than 1.5 MJ/m^2 at the control was 0.55 kPa, while the mean was 0.68 kPa, with a range in Δ from 0.08 to 2.91 kPa, showing that under these conditions the forest was always more humid than the control (Figure 5.12).

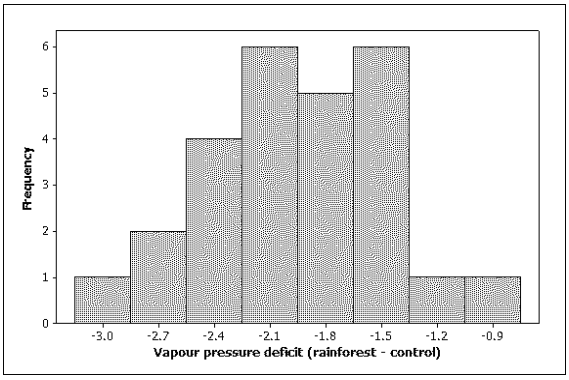

Figure 5.10 Frequency histogram of the Δ between inside and outside forest conditions for vapour pressure deficit (kPa) when temperatures are greater than 25 °C and relative humidities less than 25% at control 1, for Site Call and Control 1.

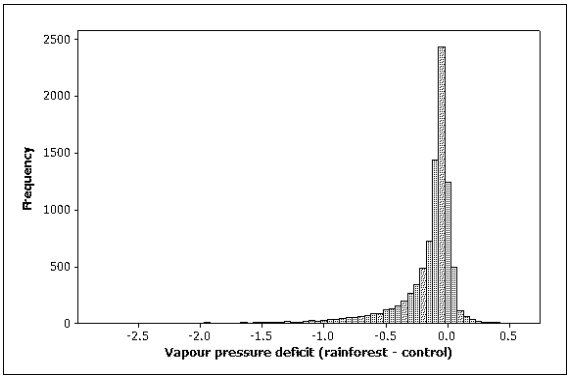

Figure 5.11 Frequency histogram of the Δ between inside and outside forest conditions for vapour pressure deficit (kPa) at Site Call and Control 1.

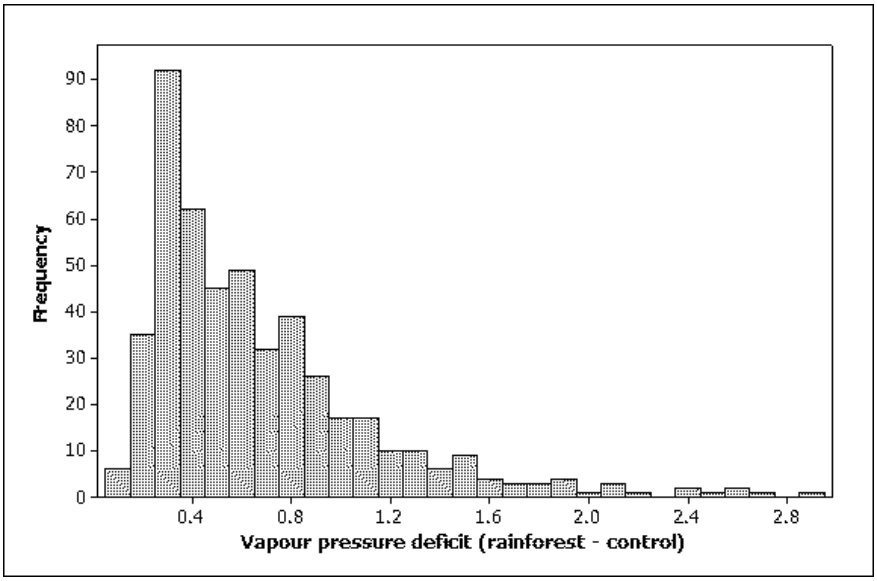

Figure 5.12 Frequency histogram of the Δ between inside and outside forest conditions for vapour pressure deficit (kPa) at Site Imp and Control 1 when the total solar radiation is greater than 1.5 MJ/m² at Control 1.

For all values of VPD at the control for Site Imp, the median Δ VPD was 0.004 kPa while the mean was 0.04 kPa, with a range from -0.77 to 0.02 kPa. Almost all values for VPD within the forest were lower than at the control, indicating that the forest was more humid (Figure 5.13). The median Δ VPD when total solar radiation was greater than 1.5 MJ/m^2 at the control was 0.125 kPa, while the mean was 0.18 kPa, with a range from -0.77 to -0.008 kPa (Figure 5.14).

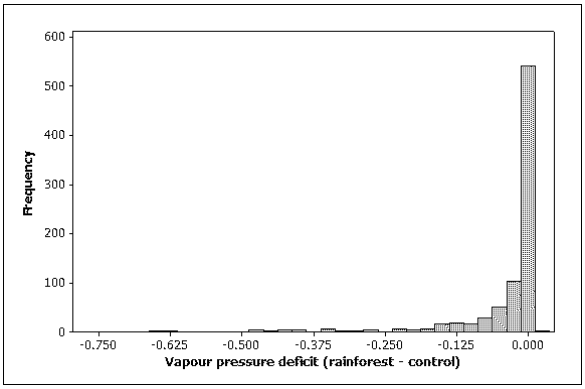

Figure 5.13 Frequency histogram of the Δ between inside and outside forest conditions for vapour pressure deficit (kPa) at Site Imp and Control 1.

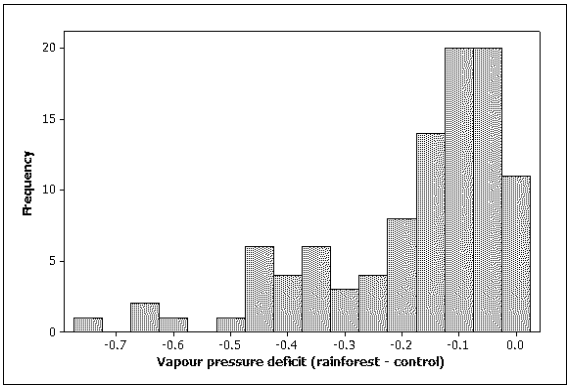

Figure 5.14 Frequency histogram of the Δ between inside and outside forest conditions for vapour pressure deficit (kPa) when the total solar radiation is greater than 1.5 MJ/m² at Control 1.

For all values of VPD at the control for Site DB, the median Δ VPD was 0.02 kPa while the mean was 0.03 kPa, with a range in Δ from -0.7 to 0.7 kPa (Figure 5.15). The median Δ VPD when total solar radiation was greater than 1.5 MJ/m^2 at the control was 0.08 kPa, while the mean was 0.09 kPa, with a range in Δ from -0.7 to 0.4 kPa (Figure 5.16). When only the summer months were included in analysis the median Δ VPD was 0.03 kPa while the mean was 0.04kPa, with a range in Δ from -0.7 to 0.7 kPa (Figure 5.17).

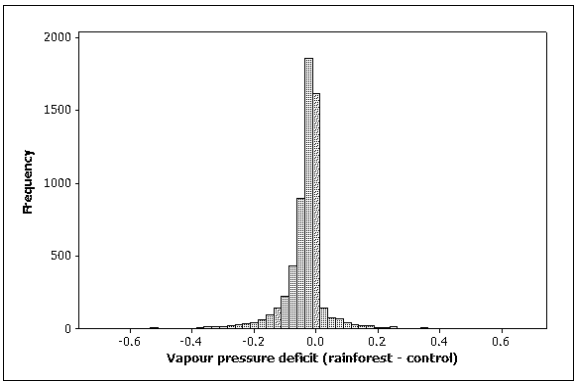

Figure 5.15 Frequency histogram of the Δ between inside and outside forest conditions for vapour pressure deficit (kPa) at Site DB and Control 2.

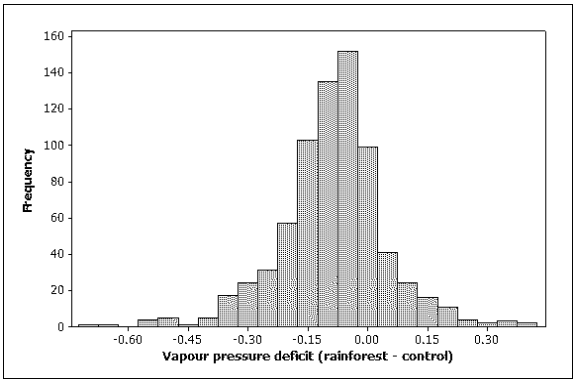

Figure 5.16 Frequency histogram of the Δ between inside and outside forest conditions for vapour pressure deficit (kPa) at Site DB and Control 2 when the total solar radiation is greater than 1.5 MJ/m² at Control 2.

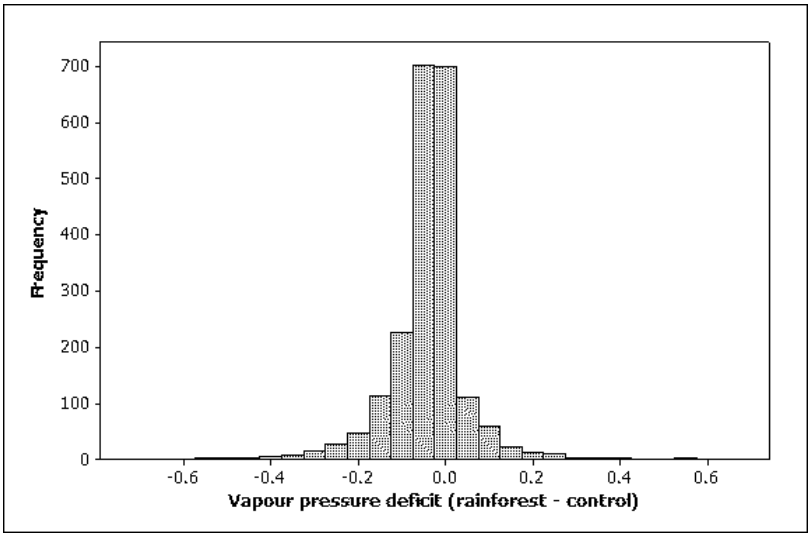

Figure 5.17 Frequency histogram of the Δ between inside and outside forest conditions for vapour pressure deficit (°C) from 1/12/2010 to 28/2/2011 for Site DB and Control 2.

5.3.3 Wind speed and direction

For all wind speeds at the control for Site Call, the median Δ wind speed was 1.02 m/s, while the mean was 1.01 m/s, with Δ wind speed ranging from -4.8 to 1.4 m/s (Figure 5.18).

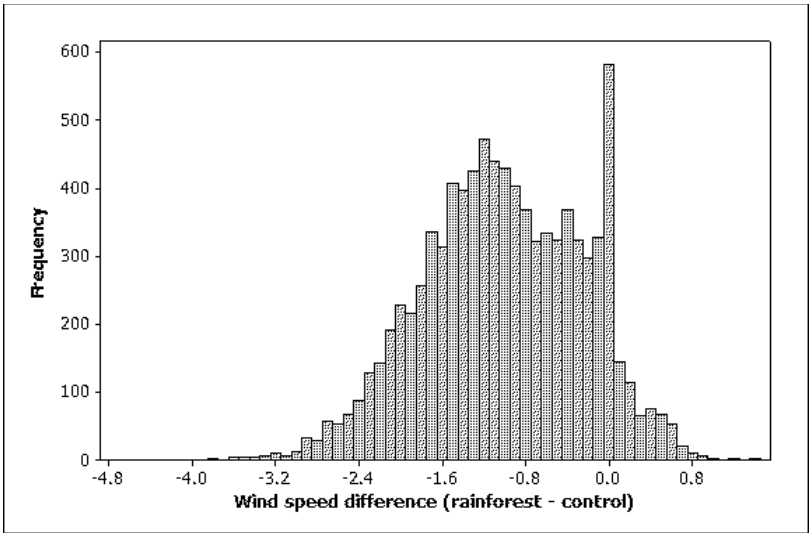

Figure 5.18 Frequency histogram of the Δ between inside and outside forest conditions for wind speeds (m/s) at Site Call and Control 1.

For all wind speeds at the control for Site DB, the median Δ wind speed was 0.68 m/s, while the mean was 0.8 m/s, with Δ wind speed ranging from -4.6 to 0.96 m/s (Figure 5.19). When only the summer months were included in analysis the median Δ wind speed was 0.73 m/s while the mean was 0.81 m/s, with a range in Δ from -3.15 to 0.79 m/s (Figure 5.20).

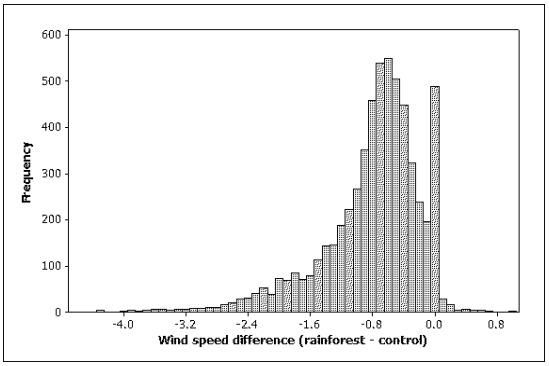

Figure 5.19 Frequency histogram of the Δ between inside and outside forest conditions for wind speeds (m/s) at Site DB and Control 2.

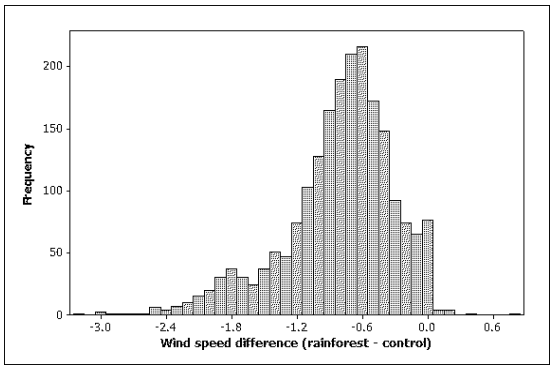

Figure 5.20 Frequency histogram of the Δ between inside and outside forest conditions for wind speeds (m/s) from 1/12/2010 to 28/2/2011 for Site DB and Control 2.

5.4 Discussion

The data captured all seasons at least once at every site, with summer data being captured three times at Site Call and Control 1. This occurred over the 2010/11 summer, the 2011/12 summer and the 2012/13 summer. The summer of 2010/11 was both wetter and cooler than average (Bureau of Meteorology 2012*a*), and although the summer of 2011/12 provided some warmer and drier conditions at the end of February, was generally cool with regular rain in the south-west (Bureau of Meteorology 2012*c*). The 2012/13 summer provided warm and dry conditions, conducive to fire weather, however, enough rain fell within south-west Tasmania that the drought conditions experienced in previous years were not present. This has somewhat constrained analysis as there was not a substantial incidence of fire weather represented in the data.

The comparison of the air temperature and relative humidity data collected from the ibutton loggers with that of the data collected from the automatic weather station suggests that the

ibutton loggers were a satisfactory substitute, particularly for recording air temperature. Relative humidity was not predicted as successfully, but the relationship was still very strong. The difference between outcomes for relative humidity and temperature may be due to the fact that the automatic weather stations recorded the maximum humidity that occurred within each hour, whereas the ibutton loggers recorded the relative humidity that occurred on each hour. If the relative humidity varied more than the air temperature within each hour, this may partially explain the poorer result.

Although other studies that predict temperatures over large spatial or temporal distances incorporate geographical variables in the derived models, this approach was not thought necessary in this study as there was minimal spatial distance between the controls and the sites, with most physical variables remaining constant. Small differences in altitude occurred between each of the forest sites and their controls and this was addressed by adjusting the air temperature at each site by the global mean environmental lapse rate (Wolfe 1964).

The differences in the interactions between Δ and the control variables between the three sites can be explained by the canopy differences between the sites. Sites Call and Imp have a much more closed canopy than Site DB, and this creates a greater buffer between the micro and macroclimates. The difference in canopy structure can be quantified in the values for Canopy Area Index (CAI), which are 3.1 at Site Call and 1.7 at Site DB, when the canopy was in full leaf. The CAI was not measured at Site Imp but would be similar, and probably greater, than at Site Call. Also, the canopy height varies greatly between the sites. Site Call has a canopy height of about 25 m, while the height of the canopy at Site Imp is about 15 m. At Site DB the canopy height is only about 5 m, with the Stevenson screen being surrounded by the lower branches of the canopy (Plate 5.1). The greater coupling between the micro and macroclimates at this site may be further exacerbated over winter as the forest is deciduous, and may explain why there is no discernible difference in Δ temperature at Site DB when temperatures are less than 0 °C. However, when only the summer months were analysed at Site DB there was almost imperceptible difference between the Δ for temperature and VPD, indicating that the presence of leaves is not increasing the influence of the forest microclimate much beyond the influence of the branch structure. The range of responses from positive to negative between Δ temperature and the control variable can be explained by the diurnal cycle, with the forest canopies moderating both day and night time temperatures.

Plate 5.1 Automatic weather station at Site DB in winter. Image date: 21 August 2010.

The median Δ temperature at Site Call was highest when only days above 25 °C and relative humidities below 25% were included in analysis. When only temperatures on sunny days were included the median Δ temperature was intermediate between the two other values. This indicates that the greatest Δ temperature between the microclimates occurs on hot summer days. This is followed by sunny days, many of which may occur in winter, with the least Δ temperature occurring when all temperatures were included in analysis. This is most likely due to the inclusion of night time temperatures, which, as explained above, moderate the Δ. These results are consistent with those of other studies. Davies-Colley *et al.* (2000) found that diurnal temperatures between a forest and pasture were much greater on sunny days, in both summer and winter and observed that the forest microclimate was generally cooler during the day, and to a lesser extent, warmer during the night. Chen *et al.* (1995) also found that the difference in air temperature between the open and forest decreased during the day but increased at night, while Chen *et al.* (1993) found that under certain weather conditions the forest had a moderating effect on temperature, but that on cloudy and rainy days there was little variation between the microclimates.

Vapour pressure deficit refers to the amount of water the air is capable of taking up before saturation occurs. Measures of VPD combined with a quantitative assessment of the forest canopy such as LAI have been found effective in predicting the understorey drying environment across a range of canopy densities, through the inverse relationship between VPD and LAI (Ray

et al. 2010). The higher VPDs at the control compared to within the forest due to the buffering effect of the forest canopy are consistent with many other studies (Chen *et al.* 1995; Davies-Colley *et al.* 2000; Ray *et al.* 2005; Ray *et al.* 2010). The correlation analysis of Δ VPD with VPD at the control for Sites Call and DB indicates that, on occasion, the control is more humid than the rainforest. This can be explained at Site Call as a result of rain starting to fall at the control, while there is a delay in rain falling in the forest due to the interception of the forest canopy (Chapter 4). This phenomenon may also be explained as a result of isolated showers falling on the control that are not widespread enough to be felt at the rainforest site (Pook *et al.* 1991; Ray *et al.* 2005). However, these explanations are less well suited to explain the results at Site DB, as the CAI is substantially lower and there is much less distance between Site DB and the control. Considering the scatter of results and low correlation coefficient apparent at Site DB for VPD, it appears that influence of canopy on VPD at Site DB is much less than for the other two sites and this is due to the less substantial buffer provided by the canopy, as a result of low CAI and reduced canopy height (Ray *et al.* 2005). Similarly to temperature, the greatest Δ VPD was observed when conditions were closest to those in which fire could carry. Similar results have been found in other studies (Chen *et al.* 1993; Davies-Colley *et al.* 2000). Under sunny conditions the greatest range in Δ VPD was observed at Site Call and the least at Site DB. This is consistent with the findings of Ray *et al.* (2005), who found that VPD was greatest the denser and taller the forest canopy was.

Wind speed and direction are determined on a global scale by differences in the atmospheric pressure resulting from variation in the amount of heat energy absorbed by the earth (Bilgili and Sahin 2010). Therefore, wind speed is independent of any of the other climatic variables collected in this study. Wind flow across the earth's surface is impacted by obstructions such as vegetation, which results in mechanical turbulence, and terrain, which affects wind flow (Gould *et al.* 2007). For Site Call it is likely that the wind speeds measured within the forest were not under the same local influences as the winds measured at the control, due to the geographical distance between the sites. On the other hand, Site DB was situated less than 200 m from its respective control, and separated by a lake, where obstruction by terrain was minimal. Studies into wind variation in a jarrah forest in Western Australia found that gusts of wind travelling into the forest decayed to 36% of their original strength in 12-20 seconds, before becoming undetectable (Gould *et al.* 2007), while Chen *et al.* (1995) found that wind speed from an edge into a forest decreased exponentially, and was influenced by external wind speed and wind direction relative to the forest edge. As the anemometer at Site Call is situated further into the

forest than at Site DB, this would further reduce the effect of wind speed recorded by the automatic weather station at Site Call. No particularly strong wind gusts were recorded at either of the sites in the present study. Gould *et al.* (2007) suggest that the predominate source of wind gusts measured within a forest are from over-canopy wind flow. This source combined with the closed nature of the rainforest canopy may explain the lack of any strong wind gusts recorded within the forest of the present study. Gould *et al.* (2007) also found that wind within the jarrah forest was highly variable, both temporally and spatially, with wind felt at one location undetectable 40 m away at the same time.

Rare examples of higher speed winds within the rainforest than at the control are likely to be due to very localised wind gusts. The greater occurrence of these winds at Site Call is probably due to the greater distance and more complex topography between the control and the rainforest. The correlation analysis of Δ wind speed with wind speed from the control shows that as wind speed increases the difference in wind speed between the rainforest and control also increases, with winds at the control becoming faster. This is in contrast to the results of Gilfedder (1988), who found that wind speeds between an open valley and nearby *Eucalyptus coccifera* woodland were much more similar when wind speeds were greater. However, Chen *et al.* (1995) found similar results to the present study, with differences between microclimates becoming greater under windy weather conditions. The old-growth Douglas-fir forests studied by Chen *et al.* (1995) are much more similar in structure to Site Call than the *Eucalyptus coccifera* woodland studied by Gilfedder (1988), and this may explain the difference in results at Site Call. However, as Site DB is more structurally similar to the *Eucalyptus coccifera* woodland studied by Gilfedder (1988) it is uncertain why Site DB does not respond in the same direction. One possible explanation is that the winds were much stronger on Mount Wellington than at the locations used in the present study. Also the open area on Mount Wellington had long fetch to the windward, whereas Lake Fenton is surrounded by forest.

It is well known that forests have a moderating effect on climatic extremes (Geiger 1965) and this is supported by the results of this study. Furthermore, the results clearly showed that density of the forest canopy affected the degree of variability in microclimate. The three forests were generally cooler and more humid than the surrounding environment, with the degree of variability in temperature and humidity being greater in the forests with a higher CAI and taller canopy. Although there appears to be a number of issues that make wind speed prediction within a forest difficult (Gould *et al.* 2007) the results show that less wind penetrates the forest than is recorded at the control sites, and that as winds increase in speed the difference in wind

speed between the forest and open areas becomes greater. In general, from the results of the present study it can be assumed that during fire weather conditions (e.g. temperature greater than 25 °C; relative humidity less than 25%) temperatures in a callidendrous forest will be about 7 °C cooler than those outside the forest, while VPD will tend to be about 2 kPa lower within the forest. During sunny conditions the temperature will be about 4 °C cooler within a callidendrous forest than an adjacent open area, about 4 °C cooler within an implicate forest and about 1 °C cooler within a deciduous beech forest. During sunny conditions VPD in a callidendrous forest will be about 0.5 kPa lower than in the open, about 0.15 kPa lower within an implicate forest and about 0.1 kPa lower within a deciduous beech forest. Wind speed within a callidendrous rainforest will be about 1 m/s slower than the wind in the open. Wind speed within a deciduous beech forest will be about 0.5 m/s slower than the open wind speed. When winds are greater than 3 m/s in the open the difference will be about 2 m/s slower within a callidendrous forest and about 1.5 m/s slower within a deciduous beech forest. These differences in microclimate between rainforest and the open and between types of rainforest have strong implications for fire management and these implications will be discussed throughout the remainder of this thesis.

Chapter 6 – Soil Dryness Index

6.1 Introduction

The Soil Dryness Index (SDI) has been in use in Tasmania for almost 40 years. Fire managers believe that the SDI is a very effective tool for indicating fire danger and fire behaviour in subhumid eucalypt-dominated vegetation of the east coast of Tasmania. However, the SDI has proven ineffective at providing reliable information on fire behaviour in the perhumid rainforest and buttongrass moorland dominated west coast (D. Taylor pers. comm. 2012). This discrepancy in effectiveness of the SDI between regions could result from a number of factors within the SDI including the unknown response of the SDI to soils that are wet for much of the time, such as organic soils; the canopy intercept factor recommended for rainforest within the SDI; the method of applying the canopy intercept factor to each vegetation type; or an under-representation of the flash run-off factor within the SDI. Other factors contributing to a poorer performance of the SDI in western Tasmania may include changes in vegetation type over time that are not updated in the calculation of the SDI; a lack of recalibration of the SDI from year to year; as well as a poor spatial representation of weather stations used to calculate the SDI in the south-west and western regions of Tasmania.

The original SDI (Mount 1972) derived the value recommended for rainforest canopy intercept from a mature pine plantation at Lidsdale State Forest, in New South Wales (see Bell & Gatenby 1969). There are many reasons why this equation is likely to be inadequate for a cool-temperate rainforest. These reasons include differences in canopy densities, leaf structure, Leaf Area Index (LAI) and the ability of *Pinus radiata* to absorb water directly through its foliage (Bell and Gatenby 1969), as well as climatic differences between regions, such as solar radiation and rainfall patterns. It is therefore likely that these factors contribute to the perceived inadequacy of the SDI in south-west and western Tasmania, where cool-temperate rainforest is dominant. Similarly, the run-off data used to calculate the SDI were collected from Lidsdale State Forest, where the soil is mineral based (Bell and Gatenby 1969) and therefore fails to capture any differences in relationships, such as in water infiltration and flash run-off, that may occur with rainfall on organic soils.

The SDI has been traditionally calculated from a number of stations around Tasmania where both rainfall and temperature data were available, or sometimes in the case of temperature, where a proxy value could be used (P. Fox-Hughes pers. comm. 2013). However, since the 2011/12 fire season the SDI values used in the calculation of the Forest Fire Danger Index (FFDI) have been derived from both 0.25° and 0.1° gridded values of temperature and precipitation (Finkele *et al.* 2006). This equates to approximately 25 x 25 km and 10 x 10 km grid cells. Thus, there are currently two different systems operating in Tasmania that are used to calculate the SDI, the point based system and the gridded system.

This chapter explores two questions. This first question examines whether the SDI in its current format adequately represents the soil dryness in rainforest. This is examined in the context of soil type; values for evapo-transpiration; and changing the canopy intercept factor designated for rainforest (Mount 1972) by using the results obtained for canopy saturation and interception in Chapter 4. The second question examines reasons for the perceived inadequacies of the SDI in western Tasmania, which include the lack of recalibration of the SDI from year to year; an inadequate flash run-off fraction; and the poor spatial representation of weather stations in south-western and western Tasmania. It is hoped that an examination of these questions will provide information on how the SDI can be used more effectively as a fire management tool in south-west and western Tasmania.

6.2 Methods

The response of the SDI to rainfall at Site Call and Site Org was examined from a soil moisture probe measuring volumetric soil moisture (VSM). The rainfall used to correspond to each site was the rain collected from the tipping bucket rain gauge at Control 1 for Site Call and the 24 hour rainfall data from the Bureau of Meteorology station on Mount Read (Bureau of Meteorology 2012*d*) for Site Org. The SDI was then calculated for both sites using the rainfall data described above and the temperature data from each of the respective controls, Control 1 and Control 3. Interception class F within the SDI was used to calculate canopy interception. The SDI at Site Call was calculated for each day between 31 January and 27 May 2012, while the SDI at Site Org was calculated for each day between 20 February and 24 April 2011. The VSM data used to correspond with the daily rainfall and SDI data was the average daily VSM collected at both Site Call and Site Org. The soil moisture probes were not calibrated for soil type,

therefore values of VSM are relative only, with zero percent VSM not necessarily corresponding to a fully dry soil profile.

A comparison between the SDI on organic and mineral soils was examined by using linear regression to test the response of VSM to rainfall at Site Call and Site Org. Linear regression was used to examine the slope of the relationships between both sites.

Pearson's product moment correlation coefficient was used to test the correlation between the Mount SDI (MSDI) and the average daily VSM at Site Call. The SDI was also recalculated for Site Call, replacing interception class F within the SDI with the canopy intercept rule determined in Chapter 4. Pearson's product moment correlation coefficient was then used to test the correlation between the SDI calculated from the Chapter 4 intercept rule and the average daily VSM at Site Call.

The Hobart evapo-transpiration tables used within the SDI were arbitrarily modified in an attempt to see if a better fit between the VSM and SDI could be achieved. This was done twice. In the first instance the evapo-transpiration values were multiplied by 0.5 (Table 6.1) and in the second instance they were multiplied by 1.25 (Table 6.2).

Table 6.1 Altered evapo-transpiration values (mm/day). Mount (1972) values multiplied by 0.5.

Feb-May	**7+**	**12+**	**17+**	**22+**	**27+**	**32+**	**37+**	**42+**
June -Jan	**5+**	**10+**	**15+**	**20+**	**25+**	**30+**	**35+**	**40+**
SDI 0+	0.5	0.75	1	1.5	2	2.5	2.75	3.25
25+	0.25	0.5	0.75	1	1.5	2	2.5	3
50+	-	0.25	0.5	1	1.25	1.75	2.25	2.5
140+	-	0.25	0.5	0.5	0.5	0.75	0.75	1
165+	-	-	-	-	0.25	0.25	0.25	0.5

Table 6.2 Altered evapo-transpiration values (mm/day). Mount (1972) values multiplied by 1.25.

Feb-May	**7+**	**12+**	**17+**	**22+**	**27+**	**32+**	**37+**	**42+**
June -Jan	**5+**	**10+**	**15+**	**20+**	**25+**	**30+**	**35+**	**40+**
SDI 0+	1.25	1.875	2.5	3.75	5	6.25	6.875	8.125
25+	0.625	1.25	1.875	2.5	3.75	5	6.25	7.5
50+	-	0.625	1.25	2.5	3.125	4.375	5.625	6.25
140+	-	0.625	1.25	1.25	1.25	1.875	1.875	2.5
165+	-	-	-	-	0.625	0.625	0.625	1.25

The SDI calculated from these tables was then correlated with the average daily VSM for Site Call using Pearson's product moment correlation coefficient. Pearson's product moment correlation coefficient was again used for the modified SDI values calculated from these tables along with the canopy intercept rule determined in Chapter 4 and the average daily VSM for Site Call.

In order to test whether evaporation has more of an effect on the SDI than transpiration the maximum daily temperature within the rainforest was used to calculate the SDI rather than the maximum daily temperature from the control site.

To quantify the differences in the SDI as a result of different canopy intercept factors the SDI was calculated from rainfall and temperature data collected from the Scotts Peak Dam automatic weather station for the period 1 January to 28 February 2010, using canopy intercept class F, to represent rainforest, and canopy intercept class B, to represent buttongrass moorland. As the SDI at the start of the calculation period was unknown an arbitrary starting value of 30 was assumed.

The daily SDI values for the Strathgordon automatic weather station were obtained for the period between 1 April 1994 and 27 June 2013.

6.3 Results

The response of soil moisture to rainfall at Site Call shows spikes in soil moisture corresponding with large rainfall events (Figure 6.1). In the periods between large rainfall events the soil moisture dropped to what appears to be an equilibrium level of around 36% with spikes above 45% occurring on three occasions (Figure 6.1).

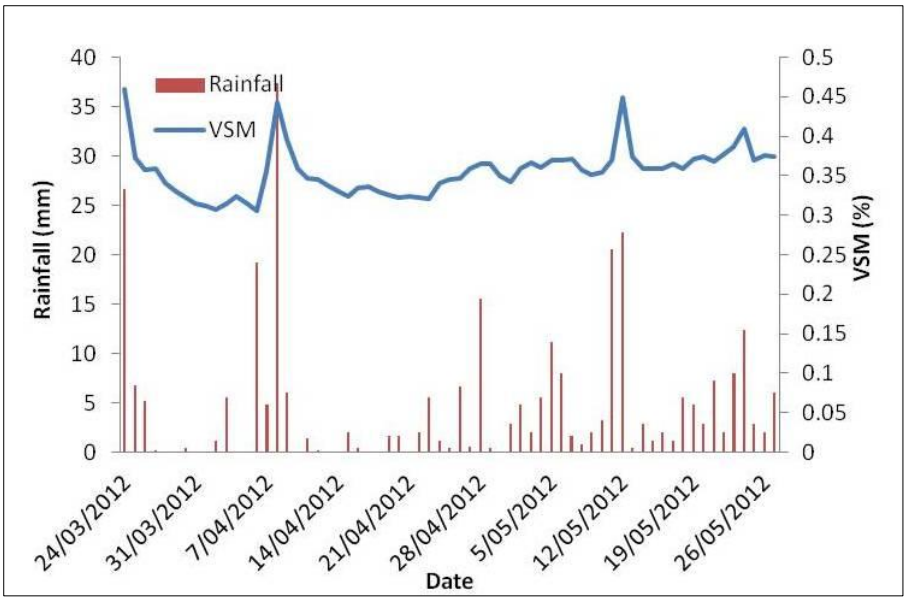

Figure 6.1 Site Call average daily volumetric soil moisture (VSM) and 24 hour rainfall (mm) until 9.00 am between 24 March and 27 May 2012.

The response of soil moisture to rainfall at Site Org shows a similar relationship between soil moisture and rainfall as observed at Site Call, with spikes in soil moisture occurring after heavy rain (Figure 6.2).

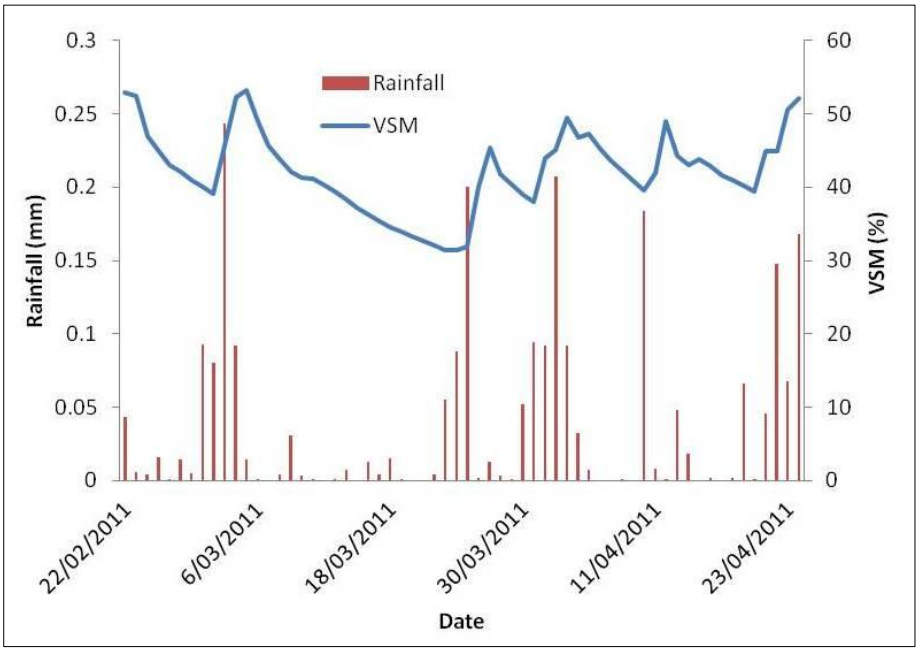

Figure 6.2 Site Org average daily volumetric soil moisture (VSM) and 24 hour rainfall (mm) until 9.00 am for Mount Read between 22 February and 24 April 2011.

At Site Call the SDI and VSM responded to each other with drops in the SDI corresponding with spikes in the VSM (Figure 6.3). The same response was observed on the organic soils at Site Org (Figure 6.4).

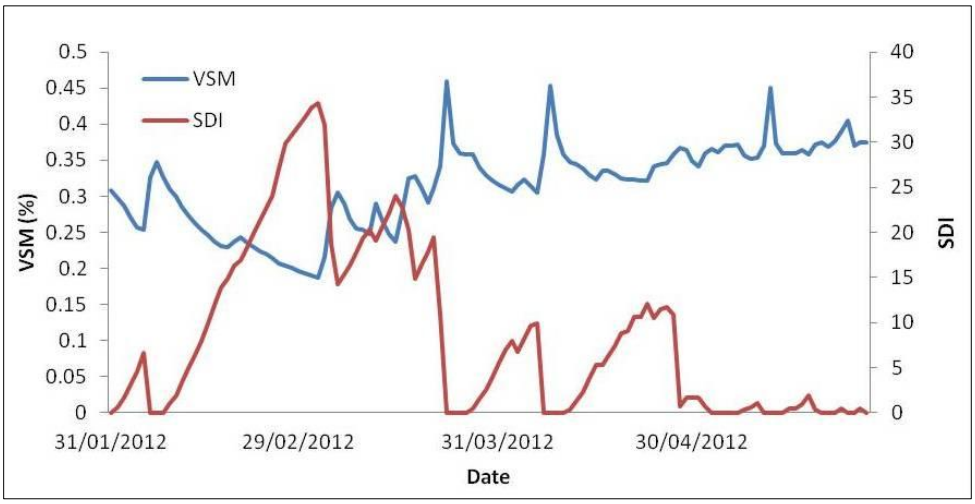

Figure 6.3 Average daily volumetric soil moisture (VSM) and daily Soil Dryness Index (SDI; intercept class F) for Site Call between 31 January and 27 May 2012.

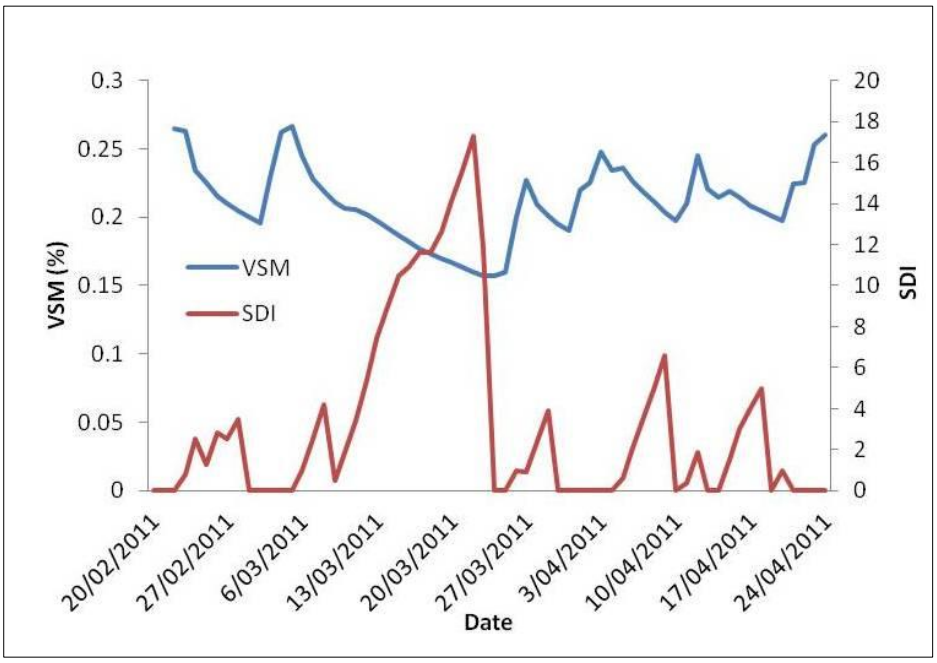

Figure 6.4 Average daily volumetric soil moisture (VSM) and daily Soil Dryness Index (SDI; intercept class F) for Site Org between 22 February and 24 April 2011.

There appeared to be no difference in the way the SDI performed on organic versus mineral soils, as the respective slopes of the linear regression lines between the VSM and rainfall for the two soil types were nearly identical (Figures 6.5 & 6.6).

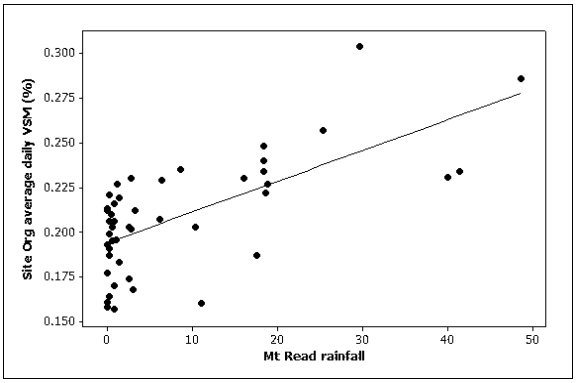

Figure 6.5 Relationship between average daily volumetric soil moisture (VSM) at Site Call and 9.00 am rainfall at Control 1 (*R*-sq = 0.22). Slope of line is 0.001781.

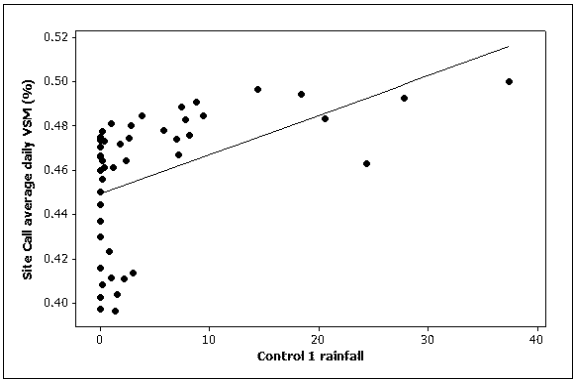

Figure 6.6 Relationship between average daily volumetric soil moisture (VSM) at Site Org and 9.00 am rainfall at Mount Read (*R*-sq = 0.47). Slope of line is 0.001724.

The correlation between the daily SDI values and the average daily VSM was -0.872; $P<0.001$ (Pearson's product moment correlation coefficient). When the canopy intercept rule determined in Chapter 4 was applied to the SDI the results were marginally weaker than those for the Mount SDI (Pearson's product moment correlation coefficient = -0.869; $P<0.001$).

The correlation between the daily Mount SDI (MSDI) with the altered evapo-transpiration figures (Table 6.1) and average daily VSM was -0.814; $P<0.001$. When the altered evapo-transpiration figures were used in conjunction with the modified interception rule the relationship was slightly stronger (Pearson's product moment correlation coefficient = -0.824; $P<0.001$).

When the evapo-transpiration tables were altered for a second time (Table 6.2) the relationship between the MSDI and the average daily VSM was -0.866; $P<0.001$. When the altered evapo-transpiration figures were used in conjunction with the modified interception rule the relationship was weaker (Pearson's product moment correlation coefficient = -0.813; $P<0.001$).

When the temperature input values for the MSDI were taken from within the rainforest the Pearson's product moment correlation coefficient was -0.805; $P<0.001$, less than when the temperature input values were taken from the control site (-0.872; $P<0.001$).

When the SDI was calculated from the same data set for interception classes B and F it was found that the SDI initially rose faster for class F than for class B and that the drops in SDI after substantial rainfall were always greater for class B than class F (Figure 6.7).

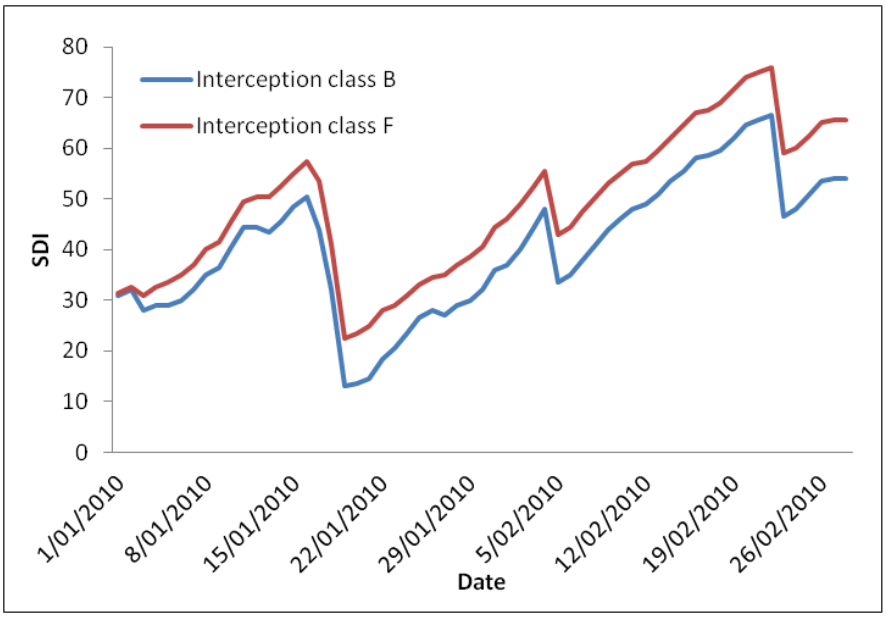

Figure 6.7 Comparison of the Soil Dryness Index (SDI) calculated for interception class B and F for Scotts Peak Dam automatic weather station from 1 January to 28 February 2010.

The SDI values for the automatic weather station at Strathgordon for the period 1 April 1994 to 27 June 2013 ranged from zero to 75. Over each winter the SDI fell to zero on a number of occasions. The highest SDI value (75) was recorded in February 1999 and the lowest summer SDI value (27) was recorded in January 2011.

6.4 Discussion

The analyses testing the adequacy of the SDI in rainforest indicate that none of the adjustments to the SDI improve the values for rainforest. Figures 6.1 and 6.2 show that soil moisture is responding to rainfall as would be expected. This response is occurring on both the mineral and organic soil sites used in the present study. Similarly, Figures 6.3 and 6.4 show that when the VSM drops there is a corresponding rise in the SDI, on both mineral and organic soils. It may be expected that the organic soils would respond differently than the mineral soils to rainfall as organic soils have a much greater water holding capacity than clay based soils, due to the large surface area of the particles and high colloid concentration (Daubenmire 1974). However, there was no apparent difference in the respective responses of the soils to additional rainfall. Figures 6.5 and 6.6 show the relative responses of the VSM to rainfall at Site Call and Site Org, and it is clear from the slopes of the lines that the responses of each site to rainfall events are very similar. It should be noted that soil moisture probes were not calibrated for soil type and therefore the VSM values on the y-axes are arbitrary and cannot be compared between sites; instead it is the response of each site to rainfall that is of importance.

The results of the manipulations of the SDI indicate that the Mount calculation is robust. Calculating the SDI by using the temperature input data from within the rainforest rather than from a nearby open location also failed to improve the correlation between SDI and VSM, indicating that the transpiration component of evapo-transpiration within rainforest plays a greater role than evaporation from the forest floor. It therefore appears that none of the changes to the SDI examined in the present study will greatly improve the effectiveness of the SDI for surface soils under rainforest vegetation.

The comparison between the SDIs using the respective intercept classes for rainforest and buttongrass showed that the interception class relating to rainforest (interception class F) always resulted in a greater SDI than when the data were calculated for buttongrass (interception class B; Figure 6.7). This means that regardless of the method used to determine the SDI for a region (point-based or gridded), in areas where vegetation is highly heterogeneous the SDI under vegetation that corresponds to a low canopy intercept factor will be underestimated and the SDI under vegetation that corresponds to a high canopy intercept factor, such as rainforest, will be overestimated. However, these discrepancies are probably impossible to avoid in any practical sense and appear to be minimal, especially when calculating the SDI from the 0.1° grid cells (Finkele *et al.* 2006). However, it is not clear whether the results found in Figure 6.7 are likely to

be a correct representation of actual soil moisture differences between these two vegetation types. No measures of soil moisture in buttongrass were conducted in the present study so it is not possible to verify if this is a correct representation of soil conditions between buttongrass and rainforest, although the personal observations of the author suggest that the opposite is true, particularly in dry conditions, which occur in the example used in Figure 6.7. The reason why intercept class F is resulting in a greater SDI value than intercept class B appears to be due to the greater interception that is accounted for in the SDI calculations. Although it is most probably true that more rain would be intercepted by a rainforest canopy than a buttongrass moorland it is also likely that there is more evaporation from a buttongrass moorland soil than from beneath the canopy of a rainforest, particularly in summer. Therefore, this seemingly erroneous outcome could result from inaccurate evapo-transpiration values. As the same evapo-transpiration data is used for all vegetation classes it is possible that the SDI could be improved by varying evapo-transpiration data by vegetation type. Although it was shown above that the transpiration component of evapo-transpiration plays a greater role within a rainforest, this may not be true for a buttongrass moorland where substantially more soil is exposed directly to sunshine. However, as the bulk density of buttongrass soil is much higher than the bulk density of rainforest soil (di Folco and Kirkpatrick 2013), evapo-transpiration from a buttongrass moorland may be slower than from rainforest, meaning that the relative soil dryness's seen in Figure 6.7 do represent reality. Thus, further research is needed to determine whether the results shown in Figure 6.7 are correct, or not.

A number of additional issues can be identified that may be resulting in the poor performance of the SDI in western Tasmania. These include changes in vegetation that may have occurred and not been updated in the application of the canopy intercept factor; the absence of any recalibrating of the SDI from year to year; an ineffective value for flash run-off; and a poor representation of weather stations used to calculate the SDI in south-western and western Tasmania.

In the case of the point based SDI values the canopy intercept classes defined by Mount (1972) that best fitted the surrounding vegetation are determined subjectively. This opinion was then corroborated by a number of other professional land managers, however once an interception class has been assigned to a station no follow up is ever conducted. This means that inappropriate intercept classes may be being applied where there have been changes in the vegetation, for example as a result of wildfire. In the case of the gridded SDI values the canopy intercept class is determined for each grid cell using the Graetz Leaf Area Index (LAI), which

has been fitted to Mount's vegetation classes that determine the canopy intercept factor (Finkele *et al.* 2006). The Graetz LAI assigned to each grid cell is based on the LAI of the dominant vegetation type (Finkele *et al.* 2006) and is not necessarily updated with changes in vegetation either.

It has been noted that the SDI rarely reaches into the hundreds in western Tasmania. Indeed, the SDI data from the Strathgordon weather station show that in 19 years the SDI never exceeded 100. This may be a result of the SDI not being recalibrated from year to year, which allows in-built errors in the calculation to accumulate over time. It is thought that these errors are due to the SDI allocating more rainfall to the soil profile than is actually entering the soil, especially over the winter months. This may be due to at least two factors. It is possible that the amount of precipitation required to return the soil profile to saturation is more than has been assumed within the SDI. It is also likely that there is an underestimation of flash run-off within the SDI calculations. The flash run-off fraction for rainforest within the SDI stipulates that 0.5 mm is lost for 18 mm of daily rainfall, 1 mm for 53 mm of daily rainfall and 1.5 mm for 88 mm of daily rainfall. It is very likely that more flash run-off than this occurs; however the amount of flash run-off will vary according to topography, soil types and vegetation communities as well as recent rainfall history and rainfall intensity (Mount 1972; Sullivan 2001), and is therefore a difficult factor to quantify.

The SDI will be recalibrated automatically during any winter when there is sufficient rain to saturate the soil profile, such as the 2010 and 2011 seasons. However, it may take only one or two drought years for the SDI to begin to under-represent the real soil moisture status in south-western Tasmania, and, if drought conditions persist for a number of years, then these errors will become cumulative. The problem of recalibration could be addressed by recalibrating the SDI at the start of each spring, to ascertain if the soil profile is as saturated as the SDI claims. The Canadian Forest Fire Weather Index (FWI) has an equivalent soil moisture calculation to the SDI, however, unlike the SDI the Canadian FWI is only calculated during the period in which there is measurable fire danger (Canadian Forestry Service 1984). This is no doubt due to the difficulty of calculating components of the FWI when there is a heavy snow cover. As a result, the FWI is recalibrated each spring and the opportunities for errors to accumulate from year to year are avoided.

Another reason why anecdotal evidence points to the SDI performing poorly in western Tasmania is the poor representation of weather stations in that part of the State. The

precipitation mapping of Nunez *et al.* (1996) indicates that the surfaces used by the Bureau of Meteorology are highly inaccurate for the southern part of western Tasmania. There are 6 weather stations from which the SDI is calculated in south-western Tasmania, and another four which border on the south-west. There are also another 6 stations along the west coast to the north of the south-west region (Table 6.3). With the exception of Hartz Mountain within the south-west, and Lake St Clair, to the north of the south-west region, all of the south-western stations are mid to low altitude sites (Table 6.3). Of these stations Dover tends to be drier than the other south-west stations and Maatsuyker Island does not appear to be representative of the other south-west stations, as significant decreases in both rainfall and mean daily minimum temperature have been documented between the decades 1979-1989 and 1990-2000 (Kirkpatrick *et al.* 2002). The station at Port Davey only collects data intermittently, while the station at Hartz Mountain is surrounded by tall vegetation, and this factor, along with being a very windy site, means that it is unlikely that this station adequately reflects the true rainfall. The distribution of weather stations along the west coast improves to the north of Strahan, although a number of these stations still collect data only intermittently.

Table 6.3 Name of weather station and its altitude used to record the Soil Dryness Index within south-west Tasmania, areas bordering south-west Tasmania, and the west coast of Tasmania.

South-west	Altitude	Bordering south-west	Altitude	West coast	Altitude
Hartz Mountain	831 m	Dover	17 m	Luncheon Hill	344 m
Maatsuyker Island	147 m	Hastings Chalet	35 m	Mount Read	1120 m
Port Davey	6 m	Lake St Clair	742 m	Rosebery	160 m
Scotts Peak	408 m	Maydena	281 m	Strahan	21 m
Strathgordon	322 m			Waratah	609 m
Warra	495 m			Zeehan	175 m

The SDI provides an effective index of soil moisture and consequently fuel dryness for rainforest. The SDI was closely associated with the corresponding VSM measurements and no alterations in the canopy interception factor for rainforest or changes in the evapo-transpiration tables increased the effectiveness of the index. Potential problems in the lack of recalibration in the SDI from year to year could be resulting in a poor performance of the SDI in western Tasmania, but the reason why this problem is not occurring in eastern Tasmania, where the chance of natural recalibration due to sufficient rainfall is less likely, is unknown. It is likely that the flash run-off factor within the SDI is under-represented, however, as flash run-off is influenced by so many different factors it is unlikely that any better values could be adopted in

place of the values determined by Mount (1972). The mountainous and topographically complex nature of the south-west would increase the variability in the SDI over short distances. This, coupled with the poor spatial representation of weather stations in the region is most likely hindering the accuracy with which the SDI can be determined in areas that are not immediately adjacent to any weather station within the south-west. The better representation of weather stations along the west coast to the north of Strahan would mean that these problems in determining the SDI in the south-west would be less in the west coast region.

This chapter has examined the effectiveness of the SDI for representing soil dryness in rainforest as well as exploring other problems that may be leading to the poor performance of the SDI in western Tasmania. The following chapter will examine how well the SDI performs as a predictor of rainforest fire in Tasmania.

Chapter 7 – Historical rainforest fires

7.1 Introduction

Fuel and weather are the two factors that most influence fire. Whereas fuel can be manipulated in order to control some aspects of fire development and spread, the weather cannot be controlled (Luke and McArthur 1978). Chapters 3 and 6 have examined some of the thresholds for fire sustainability in rainforest related to fuel dryness. This chapter examines the weather conditions associated with past fires in rainforest and adjacent to rainforest, in order to determine whether there are any clear climatic predictors for rainforest loss to fire.

7.2 Methods

7.2.1 Calculation of the Forest Fire Danger Index

The Forest Fire Danger Index (FFDI) was calculated for the first day of major fire activity for each of the past fires reviewed here. For most fires the input data was the worst fire weather conditions to occur during that day, which were generally around 3.00 pm. These conditions were assessed from the half hourly Bureau of Meteorology data for the most relevant nearby station. For some fires, however, certain input variables have had to be estimated, taken from indirect sources, or were only available for one or two times during the day and therefore do not necessarily correlate with the worst fire weather conditions of the day. The FFDI input data were daily measurements of temperature, relative humidity, 10 minute average wind speed and the Drought Factor (Appendix 1).

For each historical fire the dominant rainforest type (i.e. implicate or callidendrous) in the surrounding environment was determined either from the Tasmanian vegetation mapping product TASVEG version 2.0 (Harris and Kitchener 2005) or from other literature. For the first day of each major fire activity, the Soil Dryness Index (SDI) value for the most relevant nearby location was obtained from the Bureau of Meteorology.

7.2.2 Rainforest mapping

The total area of rainforest burned for each fire, as well as the percentage of rainforest that occurred within the boundary of each fire scar was determined using ArcMap version 10.1. The input layers were maps of fire scars, produced by the Tasmanian Parks and Wildlife Service, and TASVEG version 2.0 rainforest community groups, excluding type RLS (*Leptospermum* with rainforest scrub). No GIS data were available for the 1933/34 fire and the distribution of rainforest within the Savage River fire is likely to have changed in the three decades since the fire. In these cases, data were sourced from published literature (Marsden-Smedley (1998) for the 1933/34 fire and Barker (1991) for the Savage River fire). For the post-2005 fires it was observed that some areas classified as rainforest by TASVEG were not in fact rainforest. Pre-2005 aerial imagery was used to re-map the areas within the fire boundaries using ArcMap version 10.1. For the Harrison's Opening, Cape Sorell and Mount Franklin-Donaldson fires, pre-1992 imagery was used for rainforest mapping.

7.2.3 Data analysis

Wilcoxon rank sum tests were used to examine the relationships between rainforest and non-rainforest fires for the following variables: rainfall in the previous 10, 20, 30, 60, 100 and 365 days, SDI, Drought Factor, FFDI, temperature, relative humidity and wind speed. It was decided that the Reynolds Creek fire should be classed as an unsuccessful rainforest fire as the rainforest that did burn was largely early successional rainforest, with very little mature rainforest being affected by fire (J. Marsden-Smedley pers. comm. 2013).

7.3 Results and discussion

Of the 12 historical fires examined, 6 were considered to be unsuccessful in burning rainforest, 5 were considered to be successful, and one was considered a borderline rainforest fire (Table 7.1).

Table 7.1 Summary table of fires and Forest Fire Danger Index. L = low fire danger rating; M = moderate fire danger rating; H = high fire danger rating; VH = very high fire danger rating; S = severe fire danger rating; E = extreme fire danger rating; C = catastrophic fire danger rating. Call = callidendrous rainforest; Imp = implicate rainforest.

Fire	Rainforest burned?	FFDI	Sources
1933/34	Y	36 (VH)	Rainfall data obtained from the Cape Sorell Bureau of Meteorology (BOM) station. Temperature and humidity data sourced from Foley (1947). Wind data used to calculate FFDI was the average of the readings reported in *The Advocate* on 10 February 1934. Drought Factor was guessed based on rainfall records from Cape Sorrel for the previous two months.
Savage River 1982	Y	57 (S)	All data sourced from Barker (1991) except wind speed data which was the 3.00 pm data from the Savage River Mine BOM station.
Harrison's Opening 1993	N	6 (M)	The Soil Dryness Index and Drought Factor were for Hobart Airport BOM station. All other data were for Geeveston (Cemetery Road) BOM station.
Cape Sorell 2001	N	4 (L)	Temperature and wind data were for Cape Sorell BOM station. Soil Dryness Index, Drought Factor and relative humidity data were for Strahan aerodrome BOM station.
Mount Frankland-Donaldson 2003	N	12 (H)	All data were for Luncheon Hill BOM station.
Mount Castor 2006	N	23 (H)	All data were for Warra BOM station.
Cracroft River 2007	N	23 (H)	All data were for Warra BOM station.
Reynolds Creek 2007	Y/N	50 (S)	All data were for Scotts Peak Dam BOM station.
Heemskirk Road 2008	Y	27 (VH)	All data were for Luncheon Hill BOM station.
Kilmore East – Murrindindi 2009	Y	100+ (C)	Temperature, relative humidity, wind gust and KBDI† data sourced from *Cruz et al.* (2012). Average wind speed data sourced from Bureau of Meteorology (2009).
Lake Macintosh 2010	Y	17 (H)	Temperature, humidity and wind data were for Mount Read BOM station and temperature and humidity have been adjusted for altitude. Soil Dryness Index and Drought Factor were for Rosebery BOM station.
Giblin River 2013	N	50 (S)	All data were for Scotts Peak Dam BOM station.

† KBDI; Keetch-Byram Drought Index.

7.3.1 Individual fires

7.3.1.1 1933/34 fire

There are few data on the 1933/34 bushfires, which, together with the 1938/39 bushfires are purported to have burned large areas of rainforest in crown fires in south-western Tasmania (Marsden-Smedley 1998). The 1933/34 fire burned an extensive area from the West Coast Range and the Lower Gordon River to Southport Lagoon (Marsden-Smedley 1998), whereas the 1938/39 fire reburned much of the area around Frenchmans Cap originally burned in the 1933/34 fire, but at a much higher intensity (Marsden-Smedley 1998). It is has been estimated that 59,364 ha of rainforest burned during the 1930s in south-west Tasmania alone (Marsden-Smedley 1998). A number of fires swept the state in mid-January 1934, while on 9 February severe weather conditions fanned devastating fires over most of the state (*The Advocate* 1934). The temperature at Zeehan reached 32° C on 9 February 1934 while the relative humidity was 52% at 3.00 pm (Foley 1947). Wind speeds of 64 to 96 km/h were recorded in southern Tasmania (*The Advocate* 1934). In the months before the 1933/34 fire, there were two months when the monthly rainfall total on the west-coast was below 50 mm (Bureau of Meteorology 2012*e*). One month of rainfall below 50 mm in any 12 month period is considered to be a threshold for major rainforest fires to occur within south-west Tasmania (Marsden-Smedley 1998). The FFDI for the 1933/34 fire was estimated to be 36, assuming a Drought Factor of 9, which is a reasonable assumption considering the minimal rainfall for the previous two months recorded at Cape Sorell (Table 7.2).

7.3.1.2 Savage River fire 1982

The Savage River fire was first observed in early February 1982. The fire trickled around for a few days until 14 February when the fire flared up as the temperature reached 34° C, the relative humidity dropped to 9% and wind gusts were recorded in excess of 74 km/h (Barker 1991). The 10 minute average wind speed at 3.00 pm recorded from the Savage River Mine Bureau of Meteorology station was 33.5 km/h. The weekly SDI for Savage River was 64, the Drought Factor was 9 and an FFDI of 57 was recorded on the day of the main fire run. The fire burned a total of 47,900 ha, which included around 15,000 ha of rainforest (Barker 1991). The rainforest within the fire was not uniformly burned, with a variety of intensities and fire types, including peat fires, surface fires, canopy fires and scorching (Barker 1991). Fire from spotting embers

tended to take hold in the canopy of dead myrtle trees and spread into the surrounding rainforest (T. Blanks pers. comm. 2013). The type of rainforest burned was predominately thamnic and implicate, with a higher percentage of crown fires occurring in the implicate rainforest (Barker 1991).

7.3.1.3 Harrison's Opening fire 1993

The Harrisons Opening fire began on 13 October 1993 from an unknown cause. The temperature recorded at 3.00 pm on 13 October at Geeveston was 14 °C, while the relative humidity was 58%. The 3.00 pm wind speed was 33.5 km/h. The SDI recorded at Hobart Airport was 102, while the Drought Factor was 9.3. The 3.00 pm FFDI on 13 October 1993 was 6. The fire burned at total of 325 ha, three hectares of which was callidendrous rainforest (Table 7.2). Rainforest formed much of the eastern edge of the fire scar, and some of the western edge as well.

7.3.1.4 Cape Sorell fire 2001

The Cape Sorell fire started as a result of arson on 7 January 2001, making its main run this same day. The temperature was recorded every three hours at Cape Sorell and reached 17.8 °C at 12.00 pm, while the relative humidity, recorded at the same time at Strahan, fell to 62%. The 12.00 pm wind speed at Cape Sorell reached 31.3 km/h. The SDI recorded at Strahan Aerodrome was 30, while the Drought Factor for 7 January 2001 was 6.5. The 12.00 pm FFDI on the same day was 4. The fire burned at total of 6,200 ha, 126 ha of which consisted of small pockets of callidendrous and thamnic rainforest (Table 7.2). The fire stopped at a large rainforest boundary on its south-eastern front.

7.3.1.5 Mount Frankland-Donaldson fire 2003

The Mount Frankland-Donaldson fire was started as a result of a series of lightning strikes on 15 November 2003, and made its main run on the same day. The temperature was recorded every three hours at Luncheon Hill and reached 23.2° C at 12.00 pm, while the relative humidity fell to 36%. The 12.00 pm wind speed was 37.1 km/h. The SDI recorded at Luncheon Hill for 15

November 2003 was 26 while the Drought Factor was 6.6. The 12.00 pm FFDI for 15 November 2003 was 12. The fire burned at total of 78,000 ha, 563 ha of which was callidendrous rainforest (Table 7.2). The fire failed to burn through substantial areas of rainforest on its southern and eastern extents.

7.3.1.6 Mount Castor fire 2006

The Mount Castor fire was ignited by a lightning strike on 20 January 2006, and made its main run two days later on 22 January 2006. The temperature at Warra reached 34.5 °C with the relative humidity falling to 29%. The 10 minute average wind speed at 3.30 pm was 35 km/h and a maximum gust of 67 km/h was recorded during the day. The SDI recorded at Warra for 22 January was 36, the Drought Factor was 7.2 and the FFDI at 3.30 pm was 23. The fire burned 2,700 ha of mainly scrub and moorland vegetation, and stopped at rainforest boundaries (Table 7.2).

7.3.1.7 Cracroft River fire 2007

The Cracroft River fire was ignited by a lightning strike on 17 February 2007. It made its main run on 18 February when the temperature at Warra peaked at 32° C and the relative humidity dropped to 24%. The 10 minute average wind speed at 2.00 pm was 28 km/h with gusts of 57 km/h recorded during the day. The SDI at Warra on 18 February was 39, the Drought Factor was 7.4 and the FFDI at 2.00 pm was 23. The fire burned a little over 13,000 ha of mainly moorland vegetation (Table 7.2). The fire did not burn into rainforest, with the rainforest boundaries often defining the limits of the fire.

7.3.1.8 Reynolds Creek fire 2007

The Reynolds Creek fire started as a result of the same lightning storm that ignited the Cracroft River fire, also making its main run on 18 February 2007. The Reynolds Creek fire occurred to the west of the Cracroft River fire with the closest Bureau of Meteorology Station being at Scotts Peak Dam. The temperature at this station reached 34.5 °C and the relative humidity fell to 22%. The average 10 minute wind speed at 3.00 pm was 54 km/h and a maximum gust of 81 km/h

was recorded during the day. The SDI at Scotts Peak Dam was 48 and the Drought Factor was 7.8 The FFDI at 3.00 pm was 50.

The fire burned a little over 25,000 ha of mainly moorland vegetation and some scrub (Table 7.2). The fire also burned small areas of rainforest and mixed forest, most notably at the southern and northern boundaries of the fire. However, as stated previously the rainforest that did burn was largely early successional rainforest. There were many islands of unburned callidendrous rainforest within the fire, as well as a large stretch of rainforest that defined much of the eastern boundary of the fire.

7.3.1.9 Heemskirk Road fire 2008

The Heemskirk Road fire was an accidental ignition that began on 15 March 2008. Two days later, on 17 March, the temperature at Luncheon Hill rose to 32 °C, and the fire made its main run. The relative humidity recorded at Luncheon Hill on 17 March was 20% and the average 10 minute wind speed at 3.30 pm was 30 km/h. Gusts up to 52 km/h were recorded during the day. The SDI for Luncheon Hill on 17 March was 99 and the Drought Factor was 9.5. The FFDI at 3.30 pm was 27. The Heemskirk Road fire burned 13,700 ha of moorland, scrub and eucalypt forest, as well as areas of implicate and callidendrous rainforest (Table 7.2).

7.3.1.10 Kilmore East – Murrindindi fire 2009

The Kilmore East-Murrindindi fire started on 7 February 2009 and burned approximately 400,000 ha of Central Victoria, with some rainforest included in this area (Cruz *et al.* 2012). The conditions on 7 February were so extreme that they precipitated the introduction of the new 'catastrophic' fire danger rating. The temperature reached 46.8 °C at Melbourne airport, the relative humidity fell below 8%, and wind gusts of 93 km/h were recorded (Cruz *et al.* 2012). The Keetch-Byram Drought Index (KBDI) was 123 at Coldstream, while the Drought Factor was 10. The fire occurred at the end of a lengthy drought period where the preceding 12 year rainfall total for central Victoria was 10-20% below the 1961-1990 average. The 3.00 pm wind speed on the ground of the Kilmore East fire was between 40-55 km/h (Bureau of Meteorology 2009) while the FFDI at this time was well over 100.

7.3.1.11 Lake Mackintosh fire 2010

Lake Mackintosh is a hydro-electric dam that was created in the 1980s through the damming of the Mackintosh and Sophia Rivers. The flooding of these valleys has resulted in the drowning of large trees, the stags of which protrude above the surface of the lake (Plate 7.1). The Lake Mackintosh fire ignited on 25 January 2010 from a lightning strike. The fire made its main run on 31 January 2010 when the temperature reached 21.8 °C at Mount Read with the humidity falling to 41%. As the automatic weather station at Mount Read sits at an elevation of 1,120 m a.s.l. whereas the elevation at Lake Mackintosh is 230 m a.s.l. the temperature and humidity were adjusted by the global mean environmental lapse rate of 0.65 °C/100 m (Wolfe 1964). This has resulted in an estimated temperature of 27.7 °C and a humidity of 34%. The average 10 minute wind speed at 2.45 pm was 35 km/h and wind gusts of up to 48 km/h were recorded during the day. The Drought Factor recorded for Rosebery on 31 January 2010 was 7.4, while the SDI was 40. After adjustment for elevation the FFDI at 2.45 pm was 17. The fire was able to burn on both sides of the lake by crossing using the dead stags (Plate 7.1).

Plate 7.1 Lake Mackintosh, showing burned stags of dead trees protruding from the lake. Image date: 5 March 2010.

The fire burned over 3,300 ha, much of which was rainforest (Table 7.2). This consisted of both crown and ground fires, where the fire trickled through the litter layer (Plates 7.2 & 7.3).

Plate 7.2 Rainforest burned on the edge of Lake Mackintosh on 31 January 2010. Image date: 5 March 2010.

Plate 7.3 Rainforest understorey burned by the Lake Mackintosh fire on 31 January 2010. Image date: 5 March 2010.

7.3.1.12 Giblin River fire 2013

The Giblin River fire was ignited by a lightning strike that occurred on 3 January 2013 in the Giblin River Valley. The fire burned over 40,000 ha to the north of Payne Bay and Bathurst Harbour, making its main run on 4 February when the temperature at Scotts Peak Dam reached 35.1°C, the relative humidity dropped to 22% and a maximum wind gust of 104 km/h was recorded. The 10 minute average wind speed at 4.00 pm was 83 km/h. The SDI at Scotts Peak Dam on 4 February was 15, the Drought Factor was 4 and the FFDI at 4.00 pm was 50. The fire mainly burned buttongrass moorlands and tended to burn around or stop at scrub boundaries. It appears that no rainforest burned in this fire (Table 7.2).

Table 7.2 Percentage rainforest burned, rainfall in the previous 30 days preceding the fire, rainfall in the previous 60 days preceding the fire and Soil Dryness Index (SDI). N/A – data not available. For rainfall in the previous 10, 20, 100 and 365 days see Appendix 2.

Fire	Percentage of rainforest within fire scar (%)	Area of rainforest burned (ha)	Rainfall in previous 30 days (mm)	Rainfall in previous 60 days (mm)	SDI
1933/34	N/A	59,364†	34.1	72.6	N/A
Savage River 1982	31	15,000	43.9	102.3	64
Harrison's Opening 1993	0.9	2.9	66.4	156.8	102
Cape Sorell 2001	2	126	69.6	78.6	30
Mount Frankland-Donaldson 2003	0.7	563	115	310.2	26
Mount Castor 2006	2.3	61	167.2	332.6	36
Cracroft River 2007	0.9	122	102.8	193	39
Reynolds Creek 2007	12.5	3,170	135.4	262.4	48
Heemskirk Road 2008	12.9	1,772	47.8	66.6	99
Kilmore East – Murrindindi 2009	N/A	N/A	7.8	67.6	123 (KBDI‡)
Lake Mackintosh 2010	65	2,201	36.8	169	40
Giblin River 2013	0.06	23	144.4	249.4	15

† Value is for the total amount of rainforest estimated to have burned during the 1930s in south-west Tasmania. This mainly occurred in two fires, the 1933/34 fire and the 1938/39 fire (Marsden-Smedley 1998).

‡ Keetch-Byram Drought Index was used rather than the Soil Dryness Index.

7.3.2 Integration of historical fire data

Marsden-Smedley (1998) observed that the years in which major rainforest fires occurred in south-west Tasmania all had at least one month with less than 50 mm of precipitation preceding the fires. Analysis of the historical fires above confirms this threshold, as all the fires in which rainforest burned were preceded by a 30 day period with less than 50 mm of rain, with the exception of the Reynolds Creek fire (Table 7.2). All the fires in which rainforest did not burn were preceded by a 30 day period where the rainfalls exceeded 50 mm (Table 7.2).

The Reynolds Creek fire burned 3,170 ha of rainforest, which corresponds to 12.5% of the total area burned in the fire (Table 7.2). However, the rainforest that was burned in the Reynolds Creek fire was largely early successional rainforest, with only a small amount of mature rainforest being affected by fire (J. Marsden-Smedley pers. comm. 2013). The Reynolds Creek fire started on the same day as the Cracroft River fire, which burned 122 ha of rainforest, or just under one percent of the total area burned (Table 7.2). The FFDI calculated for Scotts Peak Dam, the nearest station to the Reynolds Creek fire, reached 50, while the FFDI at Warra, the nearest station to the Cracroft River fire, reached 23. Although these fires occurred on the same day it is clear from the data from the weather stations that the fire weather was increasingly severe further to the west, especially as a result of increased wind speeds recorded at Scotts Peak Dam. The temperature was also a few degrees warmer, the humidity a few percentage points lower and the Drought Factor marginally higher at Scotts Peak Dam (Appendix 1).

What appears to be a low FFDI value for the 1933/34 fire, considering the amount of rainforest that is purported to have burned, is due to the high relative humidity value of 52%. It is likely that this value does not correctly represent the minimum relative humidity on 9 February 1934 for Zeehan. Climate data for the 1933/34 fire were difficult to obtain as the Bureau of Meteorology does not have digitised records for temperature and relative humidity from any station on the west coast of Tasmania before 1957. The value for relative humidity was taken from Foley (1947), and only the 3.00 pm value for Zeehan on 9 February 1934 was published. Considering the temperature at Zeehan reached 32 °C that day it is likely that the relative humidity was also low, as high temperatures are not often associated with humid conditions in Tasmania (Foley 1947). It is possible that the relative humidity was only recorded at 3.00 pm, by which time a south-westerly change had already occurred, raising the humidity. If this was the case it is likely the peak fire danger conditions occurred earlier in the day, with an associated lower relative humidity. Fox-Hughes (2008) reported an FFDI of 92 in Hobart for this same date. Fox-Hughes (2008) also found that noon is the most frequent time of peak fire danger at Strahan. Considering the much higher FFDI found for Hobart for this same date and the earlier peak fire danger found to occur frequently on the west coast, it is most likely that the peak FFDI for the west coast was much higher than has been reported in Table 7.1. Based on relative humidity values and corresponding temperatures recorded for the other fires examined in this chapter, it could be reasonably estimated that the relative humidity fell to 25% during the peak fire weather conditions of the 1933/34 fire. If this were the case then the associated FFDI would have been 90, which corresponds with an FFDI of Extreme. With the exception of the Kilmore East – Murrindindi fire in Victoria this is the highest FFDI rating found for any of the fires

examined in this study and supports the large amount of rainforest that burned during this fire (Marsden-Smedley 1998).

The Lake Mackintosh fire had the largest percentage area of burned rainforest of all the fires examined (Table 7.2), however the FFDI reached only 17. This is the lowest FFDI of any of the fires studied that successfully burned rainforest, and lower than three fires that failed to burn rainforest, yet over 2,000 ha of rainforest burned in this fire. The temperature, humidity and wind speed data used to calculate the FFDI were recorded at the Mount Read weather station, which occurs at an elevation of 1,120 m a.s.l., some 900 m higher than Lake Mackintosh itself. However, as the temperature and humidity data were adjusted to take into account the global mean environmental lapse rate, the FFDI value should be close to accurate for the area around Lake Mackintosh. Indeed, as wind speed was unadjusted, and is likely to be greater at higher elevations, it is likely that the FFDI value reported may be greater than would have been recorded nearer to the fire. Although the FFDI conditions surrounding the Lake Mackintosh fire were subdued, the Lake Mackintosh fire fulfilled the criterion of receiving less than 50 mm of rain in the 30 days prior to the fire making its main run.

The Savage River fire burned the largest area of rainforest of any of the post 1930s fires, at 15,000 ha. The fire is reported to have burned mostly implicate and thamnic rainforest (Barker 1991). Implicate rainforests tend to occur in cooler and more humid environments than callidendrous rainforests and this is likely to reduce the chance of fire occurring in them as compared to callidendrous rainforests. However, the higher fuel load and more continuous fuel ladder from the forest floor to the canopy of implicate rainforests are likely to result in higher flammability than callidendrous rainforests. It is not clear how this discontinuity in the flammability of the two rainforest groups expresses itself, however, due to the cooler and more humid position in the landscape that implicate rainforests occupy it is likely that implicate rainforests would require more extreme climatic conditions and more drying to occur before they ignite, but once ignition within the implicate rainforest has occurred then the fire would burn at a higher intensity than a callidendrous rainforest, where the sparse understorey would limit the extent of fire damage. It is possible that this is the reason why such large tracts of rainforest were able to burn in the Savage River fire.

The Giblin River fire had the most extreme weather conditions of all Tasmanian fires, except the Savage River fire. However, the Giblin River fire is reported to have burned only buttongrass, with rainforest, eucalypt forest and even scrub remaining unburned. Although the temperature,

humidity and wind associated with this fire were all extreme, the Drought Factor and SDI recorded for Scotts Peak Dam were very low. It appears that the very wet nature of the soil and vegetation at the time of this fire resulted in the absence of forest being burned, and that this was a more important factor than high temperatures, low humidities and high wind speeds.

Rainfall in the previous 30 days is the strongest determinant of rainforest fire, with rainfall in the previous 20 and 60 days also being significant (Table 7.3). The only other significant variable is Drought Factor, with the other drought related variable, Soil Dryness Index, not proving significant (Table 7.3). This latter result may be because the SDI value used for one non-rainforest fire, the Harrison's Opening fire, was very high (102). This value was taken from Hobart Airport, as this was the nearest station to the Harrison's Opening fire which recorded SDI at the time, and would be expected to be much higher than SDI values to the south-west of Hobart Airport. When the Harrison's Opening fire was excluded from the Wilcoxon rank sum tests, SDI became significant (Table 7.3).

Table 7.3 Results of Wilcoxon rank sum tests for determinants of successful rainforest fires

Variable	*W*-value	*P*-value
Rainfall in previous 10 days	24.5	0.290
Rainfall in previous 20 days	31.0	**0.030**
Rainfall in previous 30 days	35.0	**0.003**
Rainfall in previous 60 days	32.0	**0.023**
Rainfall in previous 100 days	29.0	0.073
Rainfall in previous 365 days	19.0	0.876
Soil Dryness Index	4.0	0.073
Soil Dryness Index (excludes Harrison's Opening fire)	1.0	**0.019**
Drought Factor	4.5	**0.042**
Temperature	12.5	0.461
Relative humidity	23.0	0.416
Wind speed	16.5	0.935
Forest Fire Danger Index	8.0	0.149

From examination of the individual fires and the Wilcoxon rank sum tests it appears that drought related factors such as soil and fuel moisture are more important variables in determining the intensity of a wildfire, and consequently the likelihood of rainforest burning, than factors such as temperature, humidity and wind speed. The reason why the FFDIs do not appear to be a good predictor of rainforest burning may be that the importance of soil and fuel moisture are not given enough weight within the indices.

Chapter 8 – Discussion and conclusions

8.1 Introduction

The major aim of this research was to understand and evaluate the climatic parameters that allow fire to ignite and sustain in rainforest with specific aims to:

1) Provide information on when and under what conditions rainforest can be effectively used as a secure boundary in hazard-reduction burning of adjacent buttongrass moorlands, and

2) To provide decision making tools during a wildfire as to whether the conditions are suitable for fire to sustain and spread within rainforest.

The chance of fire occurring in rainforest is determined by the inherent flammability of the species, the condition of the fuel, and the weather conditions. The present study found that there were differences in the inherent flammability of different rainforest communities, for example between callidendrous and implicate rainforests and deciduous beech communities. The present study also found that fuel conditions are more important determinants of rainforest fire than weather conditions at the time of the fire.

8.1.1 Community flammability

The results of the present study suggest that there is an inherent difference in the flammability between the two broad types of rainforest examined, deciduous beech forest and implicate/callidendrous rainforest. *Nothofagus gunnii* and *Bauera rubioides* had the lowest flammability of all the leaf species examined, while species from the callidendrous and implicate rainforests, particularly *Atherosperma moschatum* and *Eucryphia lucida*, displayed characteristics consistent with being highly flammable. These results suggest that deciduous beech forests have a lower inherent flammability than callidendrous or implicate forests, influenced by the leaf structure of *N. gunnii*. On the other hand, there appeared to be little difference in foliage flammability between species found in callidendrous and implicate rainforests and species from adjacent flammable communities, such as *Eucalyptus nitida* and *Banksia marginata*. It is therefore

deduced that the very low flammability of callidendrous and implicate rainforests is a result of those communities maintaining a strong microclimatic barrier beneath their dense canopies rather than as a consequence of comparatively low foliage flammability. The deciduous beech forest understorey showed very little microclimatic decoupling from the surrounding open environment. Hence it is thought that the low flammabilities observed within these three forest types are the result of very different factors, with deciduous beech forest achieving a state of low flammability due to the sparse positioning of the leaves along the stem and the callidendrous and implicate rainforests achieving a state of low flammability as a result of maintaining a cool and humid microclimate. As microclimate appears to be a key factor in reducing the flammability of callidendrous and implicate rainforests, it is considered important to maintain the integrity of rainforest within Tasmania so that a strong microclimatic gradient between rainforest and surrounding open conditions continues to exist.

8.1.2 Predictors of rainforest fire

The condition of the fuel appears to have the greatest effect on the likelihood of rainforest burning and appears to be the best predictor of fire occurring in rainforest. Fuel condition is determined from the dryness of the soil and in Tasmania is represented in the Forest Fire Danger Index (FFDI), by the Drought Factor (DF) and the Soil Dryness Index (SDI), which is a subset of the DF. The DF was significant in determining the occurrence of rainforest fire (P = 0.042), as was the SDI when the Harrison's Opening fire outlier was excluded from the analysis (P = 0.019). However, the strongest relationship of any variable to rainforest fire was not an index but was the rainfall to occur in the 30 days prior to a fire, with rainforest fire highly probable if the total rainfall did not exceed 50 mm (P = 0.003). In contrast to the drought related variables, the prevailing weather conditions at the time of a major flare up in a wildfire did not appear to strongly determine whether a fire would burn rainforest or not. None of the immediate weather factors such as temperature, relative humidity or wind speed appeared to affect the chance of rainforest burning in any of the historical fires examined in Chapter 7.

Chapter 6 showed that the SDI was generally effective at providing a reliable index of soil dryness within rainforest. The SDI is one of the most important tools used by fire managers to provide guidelines of when hazard reduction burning can be conducted in different vegetation types. Guidelines have been determined for dry eucalypt forest, heathland and scrub as well as buttongrass moorland (Table 8.1), and although they are for use in hazard-reduction burning,

they also provide an indication of wildfire behaviour. However, there is very little knowledge of what SDI values are likely to result in fire spreading into rainforest. Of the fires examined in this study, all rainforest fires had SDIs above 50, with the exception of the Lake Mackintosh fire (40), and the Reynolds Creek fire (48), which, as discussed in Chapter 7, can be considered borderline in its determination as a successful rainforest fire. Similarly, of the non-rainforest fires, all had SDIs below 50, with the exception of the Harrison's Opening fire, discussed above.

Table 8.1 Soil Dryness Index guidelines for hazard-reduction burning in different vegetation types (from Marsden-Smedley 2009).

Vegetation type	SDI
Dry eucalypt forest	<125
Heathland and scrub	15-25
Buttongrass moorland	<10

The potential discrepancy observed in Chapter 6 between the SDI values for rainforest and buttongrass moorland is not likely to be an issue in the guidelines for hazard reduction burning as they have been worked out independently of each other and are relative to each vegetation type.

Thus, the results of the present study suggest that a wildfire is likely to spread into rainforest if the total rainfall in the previous 30 days has been less than 50 mm, the SDI is greater than 50 and the dead fuel moisture content of the rainforest litter is less than 16% (Chapter 3). However, as measurements of dead fuel moisture content have only been collected during low intensity experimental fires this value is untested for high intensity fires.

8.2 Other outcomes

There were a number of other conclusions arising from the present study, which throw new light upon aspects of the ecology of Tasmanian cool-temperate rainforest, but proved not to be relevant to the major aims of the thesis.

The results of the canopy interception study (Chapter 4) indicate that at least two millimetres of rain is required to saturate the canopy of a callidendrous rainforest, and about 20% of the

remaining rain is intercepted by the canopy. Chapter 4 also demonstrated that the distribution of rainfall within and beneath the rainforest canopy is highly heterogeneous.

Chapter 5 demonstrated that during fire weather conditions (temperature greater than 25 °C; relative humidity less than 25%) temperature in a callidendrous rainforest is about 7 °C cooler than that outside the forest, while vapour pressure deficit is about 2 kPa less within the callidendrous rainforest. It was also found that as external winds increase in speed, the difference in wind speed within a rainforest and adjoining open area becomes greater.

From the conclusions found in the present study some practical implications arise. The results from the canopy interception studies suggest that the first two millimetres of rain that falls onto a dry rainforest canopy are stored within the canopy. Although the SDI states that for the first two millimetres of rain only one millimetre is intercepted it is not recommended that the SDI be altered on this basis as tests were subsequently conducted that suggest that the canopy intercept factor corresponding to rainforest within the SDI adequately reflects reality. However, there may be scope when calculating the Drought Factor for the first two millimetres of rain falling on a dry canopy to be excluded. Forestry Tasmania already discounts daily rainfall values less than two millimetres for all vegetation types, regardless of a wet or dry canopy, when calculating the Drought Factor, but this method is not made explicit within the calculation of the FFDI (McArthur 1973). However, if the calculation of Drought Factor for rainforest vegetation was altered in the way described above, it would make the Drought Factor vegetation specific, which it currently is not.

The canopy intercept factor used to determine the SDI is not currently updated to reflect shifts that may occur in vegetation over time. Potential vegetation changes should be examined from time to time to ensure that the correct canopy intercept factor is still being applied for each station, in the case of the point based calculation, or each grid square, in the case of the gridded calculation.

A wildfire occurring in south-west or western Tasmania can be expected to burn rainforest if the rainfall in the previous month was less than 50 mm. It would follow that the same applies for hazard-reduction burns.

8.2.1 Further study

From this research a number of questions have arisen that provide scope for further research. Some of these are listed below.

There are very few studies that have examined the long-term effect of ground fires on rainforest in Tasmania (however see work by Hill (1982) and Hill and Read (1984)). In addition to these studies, observations by the author suggest that substantial damage and rainforest death occurs after ground fires, even of very low intensity. This appears to be either through ring barking (as observed by Hill (1982)), or by the underground burning of the root mass (S. Whight Pers. comm. 2013), which may be resulting in drought stress, or opening up pathways for pathogens.

Assessing the extent to which rainforest burned in each of the historical fires was problematic, considering the mapping of fire boundaries is generally done from aerial vehicles or satellite. It is therefore important to have ground-truthing of rainforest fires in order to adequately assess and quantify the extent of rainforest damage that occurs in wildfires.

The full study in to *in situ* rainforest flammability documented in Chapter 3 had to be aborted due to inappropriate weather conditions. It would be interesting to add to the data set that was collected in order that this study could be completed. It is desirable to collect more data using the same technique.

One deficiency that was observed in the SDI was the lack of different evapo-transpiration values for different vegetation types. A study into how evapo-transpiration responds within the main vegetation types of Tasmania could improve the SDI.

The SDI calculated for buttongrass indicated that the soil in buttongrass moorlands is drier than the soil in rainforest. Observations from the author suggest that the opposite is true but this should be tested, as well as examining any possible seasonal variation.

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Appendix 1 – Input variables for calculating the FFDI for each fire

Fire	Temp-erature (°C)	Relative humidity (%)	10 m wind speed (km/h)	Drought Factor	Time of readings	BOM weather station or source of data
1933/34	32	52	80	9	RH from 3.00 pm. Other data unknown.	Wind data reported in *The Advocate* (1934). Temperature and humidity data from Foley (1947).
Savage River 1982	34	9	34	9	Wind speed from 3.00 pm. Other data unknown.	Temperature, humidity and Soil Dryness Index (SDI) from Barker (1991). Wind data from Savage River Mine Bureau of Meteorology weather station.
Harrison's Opening 1993	14	58	33.5	9.3	3.00 pm.	Temperature, humidity and wind from Geeveston (Cemetery Road). SDI and Drought Factor (DF) from Hobart Airport.
Cape Sorell 2001	18	62	31	6.5	12.00 pm.	Temperature, humidity and wind from Cape Sorell automatic weather station (AWS). SDI and DF from Strahan aerodrome.
Mount Frankland-Donaldson 2003	23	36	37	6.6	12.00 pm.	Luncheon Hill AWS.
Mount Castor 2006	34	29	35	7.2	3.30 pm.	Warra AWS.
Cracroft River 2007	32	24	28	7.4	2.00 pm.	Warra AWS.
Reynolds Creek 2007	34	22	54	7.8	3.00 pm.	Scotts Peak Dam AWS.
Heemskirk Road 2008	31	26	30	9.5	3.30 pm.	Luncheon Hill AWS.
Kilmore East – Murrindindi 2009	47	8	47.5	10	Wind 3.00 pm.	Temperature and humidity data from Cruz *et al.* (2012). Wind data from Bureau of Meteorology (2009).
Lake Mackintosh 2010	28	34	35	7.4	2.45 pm.	Mount Read AWS.
Giblin River 2013	33	22	83	4	4.00 pm.	Scotts Peak AWS.

Appendix 2 – Rainfall values for different time periods preceding each fire

Fire	Rainfall					
	10 days	**20 days**	**30 days**	**60 days**	**100 days**	**365 days**
1933/34	16	30.7	34.1	72.6	127	1,100
Savage River 1982	2.4	24.6	43.9	102.3	304.2	1896.3
Harrison's Opening 1993	33.4	51.6	66.4	156.8	204.8	684.4
Cape Sorell 2001	7.6	58.8	69.6	78.6	101.85	1,030
Mount Frankland-Donaldson 2003	2	36.8	115	310.2	597.8	1,569.6
Mount Castor 2006	6.6	32.6	167.2	332.6	519.2	1,671
Cracroft River 2007	4.2	27.8	102.8	193	342.8	1,738
Reynolds Creek 2007	0.6	25.8	135.4	262.4	392	1,572
Heemskirk Road 2008	0	5.2	47.8	66.6	81.6	1,356.6
Kilmore-East Murrindindi 2009	3	7.6	7.8	66.6	168.8	551.2
Lake Mackintosh 2010	2	31.2	36.8	168	238.4	1,845.4
Giblin River 2013	39.6	107.6	144.4	249.4	468.4	2,067