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Changing Fire Regimes in Tropical and Subtropical Australia

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Abstract

The focus of the study is to investigate regional and local past, present and future changes in fire regimes of tropical and subtropical Queensland and shifts in vegetation composition and structure. Fire has been shown to be a significant driver of ecosystem evolution, composition and distribution through its impact on biota. Within Australia fire has long played a role in shaping the landscape, with increased fire frequency, associated with heightened aridity, over the last five million years promoting the expansion of fire adapted sclerophyll vegetation across the continent. Evidence of anthropogenic fires date back to approximately 50 ka (thousand years ago) with the advent of Aboriginal occupation and fire-stick practices, however with the arrival of Europeans there was a decline in fire frequencies, related to fire exclusion that observes an increase in fire intensity and severity.

A review of the introduction of tropical African perennial grasses to improve grazing in tropical and semi-arid regions of northern Australia was also undertaken. This introduction has resulted in some exotic grass species such as Gamba grass (*Andropogon gayanus*), Mission grass (*Cenchrus polystachios* syn. *Pennisetum polystachion*) and Guinea grass (*Megathyrsus maximus syn. Panicum maximum Jacq. var. trichoglume*) becoming invasive pests. Invasion by these exotic grasses has serious implications for ecosystem function, altering fire regime dynamics through increasing the distribution and abundance of fine fuels. With increased fine fuels there is a serious danger that there will be an increase in fire frequency and intensity resulting in higher severity burns and higher vegetation mortality, with possible local species extinctions and habitat modification or change.

Macro charcoal and pollen records were used from Fraser Island, subtropical eastern Australia to identify fire and vegetation histories, which show substantial temporal and spatial changes in past fire frequencies and vegetation composition for the last 24,000 years. Pollen records show pyrophobic rainforest taxa dominated and then declined while pyrogenic sclerophyll arboreal taxa increased correlating with an increase in fire frequencies, and a dryer climate. This was followed by a dramatic increase in Restionaceae values at the beginning of the Holocene (~10,000 years ago) that dropped off as a marked peak in mangroves, primarily the Rhizophoraceae and *Melaleuca*

occurred, possibly linked with sea level rise approximately 6000 to 5000 years ago, which was also associated with lower fire frequencies. Restionaceae then recovered from around 2 ka to the European settlement period, when a dramatic change in fire frequency occurred linked to fire suppression and was followed by vegetation thickening (i.e. increase in arboreal taxa) in the mid to late 20th century.

Vegetation thickening was investigated on Fraser Island through land change analysis of aerial photographs and survey data between 1958 and 2016 of a wetland system at Moon Point. This was undertaken using the Land Change Modeller (IDRISI TerrSet), with results showing that forest and woodland communities have invaded the fringes of a restiad dominated wetland. Pollen results from adjacent sediment cores support the occurrence of vegetation thickening that appears to be linked to marked changes in fire regimes on the island associated with European management since the 19th century. A projection of further landscape change was made to 2066 and this suggested a 30% loss in wetland extent by this time under present fire frequencies (i.e. with a mean of approximately one fire every plus/minus 12 years).

Identifying past and present fire regimes and vegetation composition are important for fire modelling as this provides possible scenario based probabilities for changes in fire frequency and intensity. Modelling is useful in that it provides managers with a tool to ascertain possible scenario based outcomes depending on the input values. Here the FireBGCv2 fire simulation model has been applied to an Australian context to provide a research simulation platform for exploring fire, vegetation, and climate dynamics that can be directly applied to fire management applications.

Fire has played an integral role in shaping the Australian landscape, with fire regimes driven by both climatic and anthropogenic factors during the late Quaternary period. Evidence of shifts in vegetation and fire regimes for the subtropics of Australia can be seen from pollen and charcoal analysis, with dramatic changes occurring over the past 24,000 years on Fraser Island. With the arrival and settlement of Europeans in the mid-19th century, fire regimes were once again changed that resulted in further vegetation shifts due to a fire exclusion policy. Further shifts in fire regimes can be seen in tropical northern Australia through the introduction on invasive grasses increasing fuel loads and fire frequency, resulting in a transformed landscape, perpetuating a fire grass cycle. However in the subtropics alterations in fire management have seen a reduction in fire frequency with a thickening of vegetation along the ecotone of *E. minus* wetlands and sclerophyllous forests at Moon Point on Fraser Island. The complexities of fire regimes for managers is obvious, therefore there is a need for a dynamic mechanistic fire simulation model that managers can use as a tool to project present and future fire events.

Declaration by author

This thesis *is composed of my original work, and contains* no material previously published or written by another person except where due reference has been made in the text. I have clearly stated the contribution by others to jointly-authored works that I have included in my thesis.

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Publications during candidature

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Moss, P., Tibby, J., Shapland, F., Fairfax, R., Stewart, P., Barr, C., Petherick, L., Gontz, A. and Sloss, C., 2016. Patterned fen formation and development from the Great Sandy Region, south-east Queensland, Australia. *Marine and Freshwater Research*, *67*(6), pp.816-827.

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Contributor	Statement of contribution
Philip Stewart (Candidate)	Designed the research (100%) Wrote the paper (100%) Edited the paper (95%)
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Contributions by others to the thesis

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Keywords

Fire regimes, climate change, fire modelling, resolution, ecosystem, habitat modification, fire dependency

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Table of Contents

List of Figures

List of Tables

Chapter 1 Introduction

Paleo and modern ecological studies into changing fire regimes, vegetation and climate in Australia show substantial alterations in fire events and vegetation composition occurring over time (Kershaw *et al.* 2002; Bowman *et al.* 2003; Mooney *et al.* 2011). However paleofire research shows that there is considerable uncertainty surrounding the impacts of climate change on fire regimes and vegetation (Gammage 2011; Mooney *et al.* 2011). Among these still unexplored impacts and mechanisms include regional and seasonal changes to climate and the implication of these changes on fire regimes, including impacts on vegetation dynamics and invasive grass species (Grassi *et al*. 2012; *Allen et al*. 2012a,b), suggesting the need for further research to understand the interactions between these multiple drivers of fire from the past and present (Haberle *et al*. 2010). This is critical for fire management into the future. This thesis addresses the gap in understanding past fire events and present day fire regimes through examining pollen and charcoal records in subtropical eastern Australia, analysis of the impacts of invasive grasses on fire regimes in the northern Australian tropics, characterizing vegetation thickening on Fraser Island through land change analysis and investigating future fire regimes through modelling. This chapter introduces the current literature and identifies gaps in paleofire, fire regime, climate and invasive species research. This is followed by defining the research objectives, outlining the general methodology and an overview of the thesis structure.

1.1 Background and gaps in present state on knowledge

Research has shown that the effects of single fires on flora within the vegetation's disturbance range in fire maintained ecosystems will not fundamentally change the system, however under climate change the cumulative effects of more frequent intense and severe wildfires remains mostly unquantified and largely theoretical (Paine *et al.* 1998; Turner *et al.* 2003; Stephens *et al.* 2013; Fairman *et al*. 2916; Feurdean *et al*. 2017). Studies in the United States on the Southwest, Rocky Mountains, northern Great Plains, Southeast, and Pacific coast regions show wildfires have increased in size, intensity and severity over recent decades through extreme weather and climate change (Liu *et al*. 2013). This finding suggests that altered fire regimes may also have a dramatic impact on vegetation communities in combination with rising temperatures and changes in precipitation patterns.

Fire regimes are described by Archibald *et al.* (2013) as a particular combination of fire characteristics, such as frequency, intensity, size, season, type, and extent. Where Bowman *et al.* (2016, p 3) takes the definition a step further stating that 'fire regime includes fire intensity, the time interval between fires, the spatial pattern of fires (size, shape), type of combustion (flaming versus smouldering), and the biogeochemical impacts that shape soils and vegetation'. Changes to any component within a fire regime such as an increase in fire frequency will have a direct impact on vegetation (D'Antonio and Vitousek 1992; Rositter *et al*. 2003). With climate change there is increasing concern that fire frequency and intensity will rise in the future as there is already evidence of an increase in the severity of fire weather in southern Australia (Bowman *et al*. 2012). According to Keeley and Syphard (2016) as the climate changes, vegetation change is likely to occur independently and as a result of fire-climate relationships. This is further compounded by the spread of invasive vegetation that has the potential to increase fuel loads, change fire frequency and increase fire intensity resulting in a pyrogenic shift in vegetation (Brooks *et al*. 2004).

Vegetation, in combination with climate and ignition source, plays an important role in determining fire regimes, with alterations in the type of plants growing in a region dramatically affecting fire frequency and intensity. An example of this can be seen in northern tropical Australia with the invasion of introduced grasses once thought to be useful for pastoral activities. Three grasses in particular, which are displacing native plant species, have serious implications for future fire regimes due to the increased fuel loads; Gamba grass (*Andropogon gayanus*), Mission grass (*Cenchrus polystachios* syn. *Pennisetum polystachion*) and Guinea grass (*Megathyrsus maximus* syn. *Panicum maximum Jacq. var. trichoglume*), which are becoming pests species (Lonsdale 1994; Driscoll *et al*. 2014; Douglas *et al*. 2016). Having the potential to cause dramatic changes in fire regimes through a positive feedback cycle, called a "grass-fire cycle" (Wagner and Fraterrigo 2015) by increasing the fine fuel loads of an area and increasing fire frequency and intensity (D'Antonio and Vitousek 1992; Rositter *et al*. 2003; Moon and Adams 2016; Fill *et al*. 2016; Svejcar *et al*. 2017).

Research shows that with increases in frequency and intensity of fires due to a rise in fine fuels there is also heightened plant mortality with a loss of biodiversity (Rossiter *et al*. 2003; Levick *et al*. 2015; Wang and Niu 2016) and woody biomass, however little is known of the impacts on native faunal species, which requires further research. Further physical changes to ecosystems through abiotic changes such as soil erosion and changed topography modifies the landscape resulting in transformed ecosystems that have altered composition and structure (Rossiter-Rachor *et al*. 2016). Anthropogenic induced grass invasion, climate change and changing fire regimes are complex processes that interact between multiple drivers across multiple spatial and temporal scales (Harris *et al*. 2016) requiring further study. In particular, there is limited information on how to implement burns in ecosystems that are dramatically altered by invasive species (Williams 2014), however burns still need to be implemented in altered environments.

A recent synthesis of the climate of subtropical eastern Australia (including Fraser Island) indicates it was relatively cooler during the Last Glacial Maximum (LGM) ~22 to 18 ka years ago followed by an increase in temperature during the deglacial period ~18 – 12 ka with further warming and drying ~12 ka (Petherick *et al*. 2013).The temperatures between 8 and 5 ka years ago were warmer than in the early Holocene period (Smith 2016). This warmer period is followed at ~5 ka by the onset of large scale climate variability caused by the El Niño/Southern Oscillation (ENSO) phenomena, and a trend toward drier conditions (Haug *et al*. 2001; Gontz *et al*. 2015), followed by an abrupt increase in ENSO magnitude ~3 ka years ago (Donders *et al*. 2007). At this time (~3000 years ago) palaeoenvironmental reconstructions from Fraser Island show the development of a heterogeneous and mosaic vegetation, which seems to be linked to a rise in climate variability and increased ENSO activity (Donders *et al*. 2006; Smith 2016). There is also evidence of European arrival and settlement (Moss *et al*. 2015) with distinct changes to fire management. From this it can be seen that palaeoenvironmental reconstructions for the region are complex, particularly in terms of spatial patterns, which are hindered by different temporal resolutions of available data (Smith 2016).

Changes in fire regimes can also directly impact community composition and structure with studies showing that fire can be used to manage habitat for conservation and natural resource management (Walsh *et al*. 2014) and that a reduction in fire frequency can directly impact ecosystem composition and structure by allowing vegetation thickening (i.e. expansion of woody taxa) and encroachment to occur (Sheuyange *et al*. 2005; Jurskis 2012; Jurskis and Underwood 2013; Murphy *et al*. 2014; Moss *et al*. 2016). To quantify vegetation cover change aerial photographs that are spatially identical and temporally different are ideal tools to assess gains or losses within vegetation types, as well as examine the impact of invasive species and/or ecological changes, such as vegetation thickening. However according to Walsh *et al*. (2014) there are two issues that require further research, the variation in fire regimes and the need to record individual fires (i.e. both prescribed and unplanned) and the second is to understand patchiness of the burnt landscape.

Modelling provides a tool to include climate change into future fire scenarios proactively to ascertain possible prescribed fire outcomes and wildfire impacts. The FireBGCv2 Landscape Fire Succession Model is such a tool. Identifying possible management prescribed burn outcomes is important and especially when combined with climate change, invasive species and increased fuel loads, which can all interact to transform ecosystems (Wardell-Johnston *et al*. 2015). Projections of future climate-driven changes in fire activity and impacts in the Northern Rockies have been developed using the FireBGCv2 model where it was found that larger, more frequent fires and changes in vegetation composition and structure with warming future climates may occur (Riley and Loehman 2016).

Contextual information about contemporary fire regimes has been provided from historical analysis of charcoal from sediment (Hantson *et al*. 2016) however integration between paleoecology and ecological modeling is needed to understand climate-vegetation-human-fire linkages (Iglesias *et al*. 2014). Macro charcoal analysis is not without its limitations, as for example charcoal fragments are derived from wood that has undergone incomplete combustion, and any successive fires are likely to have burnt charcoal deposits in the soil, thus removing records of previous fire events at the local scale (Waito *et al.* 2015) Fire intensity will also have potential effects of differential wood combustion on the different taxon (Godwin and Tansley 1941).

1.2 Problem Statement and Research Objectives

Fire regimes are impacted by a range of complex factors, including natural processes and human impacts and this study will investigate the fire regimes of tropical and subtropical Australia through examining past, present and future fire regimes in this region, particularly the relative influence of climate change and human influence. The key research objectives of this study is listed below.

1. Review fire regime and fire histories of South East Queensland to identify past fire regimes of the region and associated plant communities.
2. Reconstruct past fire histories and fire regimes with associated vegetation communities in the landscape from the Last Glacial Maximum to the late Holocene to provide context to contemporary fire management.
3. Identify changes to fire regimes and natural ecosystems through the invasion of exotic grass species and increased fuel loads.
4. Provide data on present day fire regimes with the associated plant communities and climate for comparison with the Paleofire and Paleoclimate data to identify any shifts or changes in wildland fire regime, climate and plant community structure.
5. Identify and provide methods to project the past and present fire regimes and climate to identify possible future fire regimes, climate and plant community structure and any shifts in plant communities using a fire.

1.3 General methodology

Macro charcoal analysis from terrestrial and lacustrine sediment cores utilising the modified method according to Stevenson and Haberle (2005) is used for the analysis of past fire regimes, and plant community structure, distribution and abundance. Present observational and recorded fire regime data available is used in combination with the paleofire data to assess changes to fire regime from the LGM and late Holocene to present and associated plant community structure, abundance and distribution. Within the analysis changes to fire frequency between late Holocene to present is assessed as suggested by Collins *et al.* (2013) that a reduction in low intensity fire has resulted in the increase in high intensity fire, which has impacted ecosystem processes and resulted in transformed ecosystems due to invasion and increased fuel loads. Paleofire and present fire data is used to forecast possible future fire regimes and ecosystem composition, structure and density changes using fire models

Project analysis tools for the statistical analysis is undertaken using the following software and analysis tools:

I. Macro Charcoal Analysis – A modified technique used by the Department of Archaeology and Natural History, Australian National University. The method has been adapted from Rhodes, A. N. method which he developed in 1995. This method has been designed for continuously sampled sediments where for example a core may be sampled along its full length from the surface to its base. It is a fast and low cost method that allows for the assessment of the frequency of charcoal peak events, providing a window into past fire regimes (Stevens and Haberle 2005).

II. CharAnalysis – is a set of diagnostic and analytical tools for analysing sediment charcoal records for the reconstruction of local fire history. The programme has been written for and functions best in MATLAB, which requires the charcoal dataset and analysis parameters to be in .xls files. In each step of the analysis process the user makes one or more parameter decisions, which must be made before the user can run the programme (Higuera 2009).

III. Review of impacts of ecosystem transformation through increased fire frequency and intensity due to invasion by invasive grasses and increased fine fuel loads.

IV. Comparative analysis using two methodologies to identify vegetation thickening and encroachment. 1) Quantitative analysis of vegetation thickening using quantified vegetation assessment methods of temporally different however spatially identical aerial photographs and, 2) IDRISI – TerrSet (V.18) Land Change Modeler to model vegetation thickening and encroachment through gains and losses and model future land change scenarios to year 2066.

V. FireBGCv2 Landscape Fire Succession Model – A research simulation platform for exploring fire and vegetation dynamics. Fire management and climate change are important factors for modern landscape management and as such new means are needed to address these challenges. Field studies, while preferable and reliable, are problematic because of the large time and space scales involved. Therefore, landscape simulation modeling has more of a role in wildland fire management as field studies become untenable. FireBGCv2 is a C++ computer programme, which is intended as a research tool (Keane *et al*. 2011) and may be used as a predictor of change.

1.4 Thesis structure

This thesis is comprised of a series of chapters. Chapter 2 reviews the current fire ecology literature from tropical to eastern subtropical regions in Australia. Chapter 3 examines the relationships between African invasive grasses and ecosystem transformation through changing fire regimes with increased fire frequency and intensity due to increases in fuel loads is discussed, providing new insight into impacts of invasive grasses. Chapter 4 reconstructs the palaeoenvironments from fossil pollen and charcoal records of Moon Point, Fraser Island. Chapter 5 researches a changing fire regime with reduced fire frequency, vegetation thickening and encroachment through modelling land change and quantitative analysis. Chapter 6 provides data and input into FireBGCv2 landscape fire simulation model and Chapter 7 assesses the outcomes of the project, identifies limitations and suggests future research.

Chapter 2. Fire Patterns of South Eastern Queensland in a Global Context: A Review

Abstract

Fire is an important driver in ecosystem evolution, composition, structure and distribution, and is vital for maintaining the environments of the Great Sandy Region (GSR). Charcoal records for the area dating back over 40, 000 years provide evidence of the great changes in vegetation composition, distribution and abundance in the region over time as a result of fire. Fires have shaped landscapes and ecosystems, creating fire-dependencies and fire disturbance-adapted flora and fauna with traits to survive fire, such as resprouting post-fire, fire ephemerals, cryptophytes and serotiny of cones and fruit. Within the region there are four main categories of fire-tolerant responses in plants and these are according to Atwell *et al* (1999), (a) the stimulation of seed release from woody capsules by heat and desiccation after fire as found in *Casuarina*, *Hakea*, *Banksia*, *Leptospermum* and *Eucalyptus*, (b) the germination of *Acacia spp.* soil-stored seed after fire, (c) sprouting from lignotubers and epicormic buds of *Eucalyptus* and *Banksia spp.* after fire and (d) the stimulation of flowering after fire.by *Xanthorrhoea spp.* However such traits are not necessarily only developed as a result of fire as a process of natural selection, other factors may play a role in such trait development within plants. Paleo-records and modern observations show a definitive link between fire, vegetation and climate, with a rise in fire with increasing temperatures. This has serious implications in a warmer world as there will be an increase in wildfire risk. Of importance is the understanding of the interactions between multiple drivers of fire regimes from the past to the present. This is critical for developing fire regime management protocols for the Great Sandy Region and other similar fire-prone regions into the future through the provision of information on climate, fire and vegetation drivers to natural resource managers.

2.1 Introduction

This paper provides a review of the global, Australian, and local context for changing fire regimes for the Great Sandy Region (GSR). This will provide important contextual information into undergoing research on the fire regimes for the past, present and future that will be vital for sustainable fire management plans for this significant region. Changes to fire regimes over time and space, as well as inking these to changes in climate, are discussed with a focus on Australia and the GSR locality. The emphasis of this paper is on shifts in fire regime over space and time and impacts that changes in climate have had on fire regime history. Linking past with present fire regimes is an integral part of the discussion, with an overarching view to look at possible drivers and changes to future fire regimes with a shifting tropical belt that is an important mechanism for future climate change, particularly fire management protocols.

The GSR is an iconic landscape, including the largest sand island in the world (Fraser Island), as well as the significant mainland Cooloola sand mass. The importance of this region has been highlighted by a number of Queensland, Australian, and international environmental protection agreements, including World Heritage and the Convention on Wetlands of International Importance (Ramsar) listings for Fraser Island, protection for the Great Sandy Strait between Fraser Island and the northern parts of the Cooloola region and National Park status for most of the GSR (Figure 2.1).

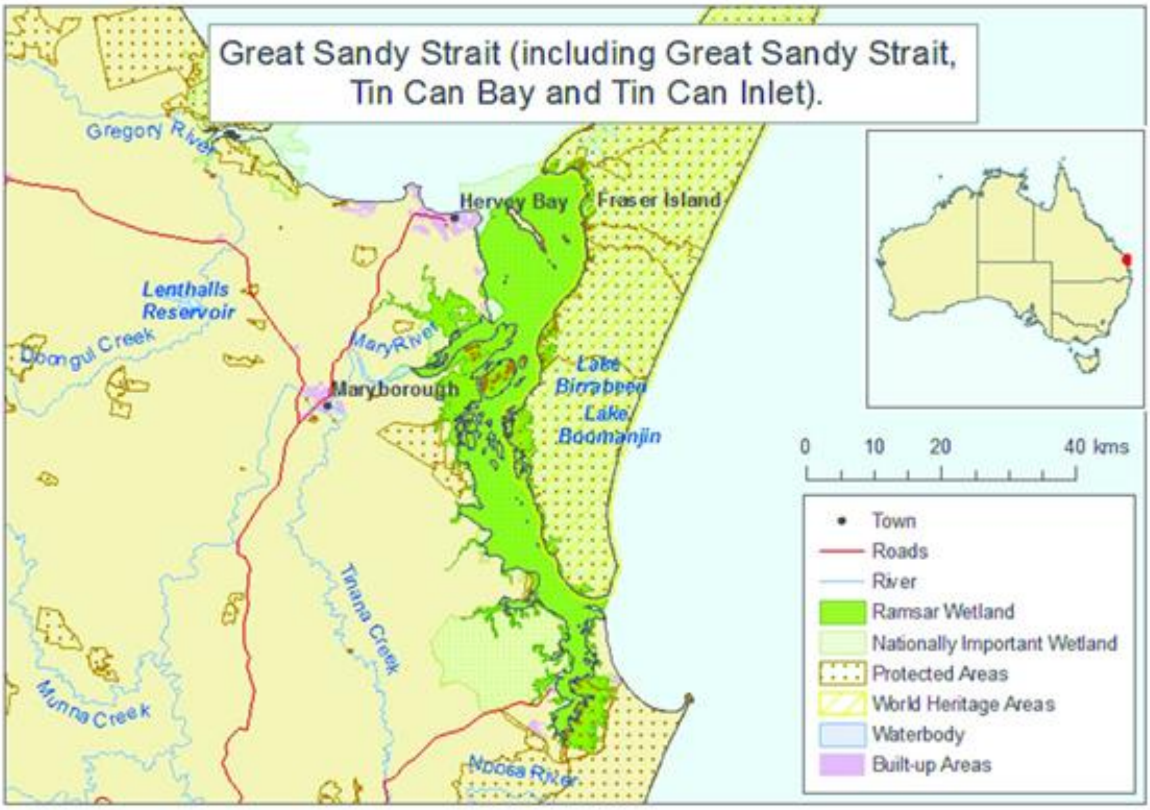


Figure 2.1. Map of the Great Sandy Strait Region, showing the major protected areas (i.e. Fraser Island National Park and Cooloola Recreation Area), Ramsar listed wetlands and World Heritage listed areas (Fraser Island). Source: http://www.environment.gov.au/cgi-bin/wetlands/ramsardetails.pl?refcode=51.

2.2 Climate and wildfire

There is strong evidence that the atmosphere is warming and that there is an increase in warm El Niño Southern Oscillation (ENSO) years with a weakening of the Walker circulation over the past decades (Knopf and Petoukhov 2007; Meng *et al.* 2012). Present data show that there has been an increase in width of the tropical belt of between 2º to 4.8º latitude north/south over the past 30 years (Seidel and Randel 2007; Grassi *et al.* 2012; Davis and Rosenlof 2012; Allen *et al*. 2012a b). This expansion has seen the shift in the subtropical subsidence, leading to an enhanced mid-latitude tropospheric warming and poleward shift of the subtropical dry zone, jet streams, and storm tracks, which is expected to contribute to an increased frequency of mid-latitude droughts in both hemispheres (Seidel *et al*. 2008; Grassi *et al*. 2012; Wilcox *et al*. 2012; Allen *et al.* 2012a; Cai *et al.* 2012).

Evidence of fire on Earth has been recorded in charcoal deposits dating back to 440 million years ago (mya) when the first stomata-bearing plants evolved (Pausas and Keeley 2009). These fires were thought to have been low intensity fires, probably due to the limited biomass available to burn and reduced atmospheric oxygen. However according to Pausas and Keeley (2009), charcoal deposits increased substantially around 345 mya during the Carboniferous Period, which correlates with the increase in atmospheric oxygen levels of the oxygen maxima through to around 2.5 mya when early hominids started evolving and using fire. In northeast Queensland, records of Aboriginal fire have been recorded between 45 and 40 ka up until the arrival of Europeans approximately 200 years ago when charcoal deposits possibly increased due to land clearing and modification of the landscape (Singh *et* al. 1981;Turney *et al.* 2001; Rule *et al.* 2012).

The south eastern Queensland region of eastern Australia (incorporating the GSR) has had limited research into past fire regimes and the links to human occupation. Direct archaeological evidence traces human occupation to around 20,000 years ago at the Wallen Wallen Creek site on North Stradbroke Island (Neal and Stock 1986) and longer charcoal records (extending back to 45,000 years) from North Stradbroke Island suggests a complex fire history (Moss *et al.* 2013). The Native Companion Lagoon site, located on the west coast of the island and within 5 km of the Wallen-Wallen site, suggests higher fire frequencies from 45,000 to 40,000 years ago, which may be analogous to the North Queensland findings. However, a similar aged record from the high dunes of the island at Tortoise Lagoon, finds no evidence of increased burning during this time, with charcoal peaks apparently related to dry climatic phases in the record. This suggests that site characteristics and local to sub-regional factors may play a key role in charcoal records and multiple sites may be required to fully understand past fire regimes for a region (Moss *et al*. 2013). More recent charcoal records from North Stradbroke Island at Myora Springs may reflect the influence of climate change and people on past fire regimes (Moss *et al*. 2013).

A shift to a warm ENSO period from between 800 to 500 years ago caused dryer conditions to prevail on the island, resulting in drought and higher anthropogenic fire frequency during this period. This was followed by cooler, wetter conditions developing around 500 years ago which correlates with a decline in charcoal deposits (Moss *et al.* 2011). With the arrival of Europeans fire regimes once again changed around 200 years ago, resulting in more frequent fires and burn-offs for agricultural, residential and industrial purposes. These changes led to changes in the landscape to fire-tolerant vegetation, *Eucalyptus* forest and *Melaleuca* swamp paperbark at the expense of less fire-tolerant species like Casuarinaceae (Moss *et al.* 2011). Of critical importance is an understanding of these interactions between multiple drivers of fire regimes from the past and present which is critical for developing fire regime management protocols for Australia in the future (Haberle *et al.* 2010).

According to Seidel *et al.* (2007) an unequivocal sign of climate change has been the warming of the atmosphere and oceans, with the thawing of land and melting of ice at the poles, and the expansion of the tropical belt. Several studies (Hu and Fu 2007; Seidel *et al.* 2008; Lu *et al.* 2009) have looked at the expansion of the tropical belt and poleward migration of the divergent air of the Hadley cell, which has important implications for subtropical populations due to possible changes in the global climate. This suggests that a shift in precipitation in the temperate regions of the planet will have profound impacts on fire regimes, ecosystems, plant communities, crop production and agriculture (Spratt and Sutton 2008; Nitschke and Innes 2013). Further, Lucas *et al.* (2014) states that recent decadal observations suggest changes in the hydro-climate with an increase in the frequency of droughts and reduced precipitation as has been observed in southern portions of Australia. This is supported by research undertaken by Moss *et al.* (2012a) suggesting that natural climate factors can significantly alter fire frequency and intensity, such as increased climate variability associated with the ENSO phenomenon that dramatically impacts fire regimes.

Wildfires have increased in size and intensity over recent decades that may be linked to extreme weather events such as extended droughts (Liu *et al.* 2013; Westerling *et al.* 2006) and where frequent low intensity surface fires in forest ecosystems have transitioned to severe fire behaviour due to climate change (Bigio 2013). These recent increases in large fires globally have caused concern about the influence climate change and humans will have on future fire regimes, with predictions of shifts to climatically driven fire for the 21st century (Pechony and Shindell 2010). Pastro *et al.* (2011), states that more intense and frequent wildfires are predicted in the future due to climate change. A better understanding of climate and wildfire interactions will be required to mitigate impacts of future fire regimes.

Collins (2014) shows a trend for a strong increase in the occurrence of high to extreme fire weather since the mid-1990s across the northern Sierra Nevada, USA. In addition to this De Groote *et al.* (2013) reviewed recent studies using several General Circulation Models that show a general increase in future fire weather severity with increased fire season in many regions around the globe. Added to this is the suggestion by McKenzie *et al.* (2014) that smoke from wildfires in the future may be more intense and widespread, further increasing the effects of greenhouse gases in the atmosphere by exerting positive radiative forcing within the short term (Simmonds *et al*. 2005; McKenzie *et al*. 2014). Liu *et al.* (2014) state that smoke particles reduce solar radiation absorbed by the Earth’s atmosphere during individual fire events that lead to regional climate effects such as reducing the surface temperature, the suppressing of cloud formation and precipitation and the enhancement of climate anomalies such as drought.

Present forecasts of climate change indicate an increase in temperatures, with warmer springs and longer fire seasons in the northern hemisphere (Westerling *et al*. 2006; Flannigan *et al*. 2013). Any increase in wildfire extent and intensity would affect human safety, livelihoods, landscape biodiversity and vegetation communities (Litschert *et al.*

2012; Rogstad *et al.* 2012). Measuring the rates of change (changes in frequency, intensity, season and sequence) in wildland fire and understanding the mechanisms responsible for such changes are important research goals (Hessl 2011). A number of modeling studies project increases in wildfire activity in future decades and empirical studies have documented change in modern fire regimes such as the development of large and intense fires under hot, dry and windy conditions in parts of the Western Desert of Western Australia where fuel loads were allowed to increase after regular traditional burning stopped in spinifex grasslands following the departure of Aboriginal people (Burrows *et al*. 2009). Research on prescribed burning in the southwestern forests of Australia by Burrows and McCaw (2013) show that fire management will require greater understanding of fuel dynamics, fire behaviour and ecosystem responses to fire in the future due to climate change.

In a study by Nitschke and Innes (2013) a statistically significant increase in mean fire (increase in the average size of wildfire) size was predicted to occur along with an increase in maximum fire size and decrease in the mean fire interval in Cordilleran forests of South-Central Interior, British Columbia. Changes in fire regimes will have direct implications for ecosystem management as the combination of large, flammable fuel types and fire-prone climatic conditions increase the risk of larger more frequent fires (Nitschke and Innes 2013; Loepfe 2012), with greater impacts on biodiversity and plant communities with increasing wildfire intensities (Beck *et al.* 2011; Bigio 2013).

2.3 Fire history and evolutionary response

Originally climate and soil were thought to control the distribution of ecosystems with fire only playing a limited role (Pausas and Keeley 2009). However fire is and has been an important driver in ecosystem evolution, composition and distribution through its impact on biota (Bond *et al.* 2005; Pausas and Keeley 2009; Coneder *et al.* 2009; Bowman *et al.* 2011; Batllori *et al.* 2013). Fire has played a key role in the evolution of plant community composition, distribution and abundance within ecosystems from around 440 million years ago (mya) when the first stomata plants evolved on the planet (Glasspool *et al.* 2004; Bond and Keeley 2005; Pausas and Keeley 2009) to present. Further charcoal records show that there has been continuous fire activity from the late Silurian to present, a period of approximately 400 million years according to Bowman *et al.* (2011).

Fire-modified ecosystems have resulted over time and space in biomes becoming fire dependent (Bond *et al.* 2005; Conedera *et al.* 2009) where fire is a major evolutionary force creating fire disturbance-adapted ecosystems. Bowman *et al.* (2011) state that fire has been an important factor in plant evolution, shaping some biomes such as the flammable savanna communities of the late Cenozoic Era. Ecosystem response and adaptation to fire has been shown to evolve over short evolutionary timescales as suggested by Litsios *et al.* (2014) with fire response traits impacting plant evolution (Wells 1969) resulting in diversity increase. However Litsios *et al.* (2014) states that this is still controversial, providing examples where rates of diversification are not associated with fire response traits for five plant genera found in fire prone habitats.

According to Conedera *et al.* (2009) wildfire is a key factor in changing the climate as it affects the chemistry of the atmosphere and carbon cycle. Enright *et al*. (2011) suggest that due to the importance of fire as an ecosystem modifier, plants developed traits to survive fire, which can be divided into two broad groups, resprouters and non-resprouters (obligate seeders). Resprouters survive fire by resprouting from fleshy below ground organs, lignotubers or rhizome, epicormics buds or apical buds whereas obligate seeders die post fire if they receive 100% leaf scorch or stem girdling, regenerating from seed through serotiny or soil stored seed (Burrows and Wardell-Johnson 2003). However such traits are not necessarily only developed through fire as a process of natural selection, other factors may play a role in such trait development within plants (Bowman *et al.* 2009).

According to Power *et al.* (2008) fire has varied globally on a continuous basis since the last glacial maximum (LGM) between 22-19 ka (Adams-Hosking *et al.* 2011) in response to changes in climate, vegetation and human land use (Bowman *et al.* 2011). Fuel availability and weather conditions are important drivers of wildfire and can be both directly and indirectly linked to climate (Duguy *et al.* 2013) vegetation type and human activity. Further, with the domestication of fire, humans substantially changed the natural fire regimes (Conedera *et al.* 2009), which in turn has changed ecosystem constituent species. Originally anthropogenic burning was thought to have created the flammable biomes, and to a certain extent this is true, however natural fires occurred millions of years before humans existed on the planet (Bond and Keeley 2005). At the biome level the variation in fire regime and biomass burnt changes with a changing climate, fuels and ignition sources (Pyne *et al.* 1996; Power *et al.* 2008).

Daniau *et al.* (2012) state that paleo-records and modern observations show a definitive link between fire and climate (temperature and precipitation) with an increase in fire with increasing temperatures. This has serious implications as in a warmer world there will be an increase in wildfire risk (Daniau *et al.* 2012). Daniau *et al.* (2012) further suggests that increases in wildfire due to increased temperature can be off-set by precipitation; however this will depend on whether or not there is an increase or decrease in precipitation at a regional level.

Fuel-limited regions of today could see increased wildfire with increased precipitation, especially with the expansion of the tropical belt and monsoons associated with future warming (Meehl *et al.* 2007; Daniau *et al.* 2012). Associated with the expansion of the tropical belt are increases in drought in the shifting subtropical dry regions (Seidel and Randel 2007) and enhanced risk of wildfire. However according to Daniau *et al.* (2012) any increase in fire due to increases in temperature for example maybe offset by changes in fuel dynamics due to changes in precipitation. The global incidence of wildfire decreased during the twentieth century despite increasing temperatures, which could be linked to landscape fragmentation and fire exclusion (Marlon *et al.* 2008; Daniau *et al.* 2012; Marlon *et al.* 2013; McWethy *et al.* 2013; Buma *et al.* 2013). However it has been suggested that the level of landscape fragmentation is no longer an effective means of wildfire suppression due to the evidence of an increase in wildfire due to natural and anthropogenic change (Mickler *et al.* 2013) and in particular the increases in wildfire being linked to global warming (Daniau *et al.* 2012).

It is well documented that wildfire regimes have been affected by climate change in the past and that future fire regimes will be driven by changes in future climates, with longer wildfire seasons, more frequent, severe and larger wildfires (Duguy *et al.* 2013; Brotons *et al.* 2013). According to Power *et al.* (2008) regional climate change has directly affected ignition and fire spread while indirectly affecting vegetation composition and structure and available fuel loads. Pechony *et al.* (2010) state that there has been an increase in concern about the influence of climate change and human activity on future wildfire and that comparatively little is actually known about the importance of these factors on wildfire. Such changes would have significant impacts on species abundance, distribution, ecosystem dynamics, and on both atmospheric and biogeochemical cycles, impacting the climate (Zinck *et al.* 2011).

Fire regimes are complex where nature and human factors can influence fire. Humans have been an integral part of nature within the Australian landscape for thousands of years playing an important role in the making of flammable Australia (Bowman *et al*. 2003). Fire regimes incorporate fire interval, intensity, season, frequency, sequence of fires and the temporal, spatial fire history and patch mosaic dynamics (Parr and Andersen 2006). Fire regimes are used to manage ecosystems to achieve conservation goals by focusing on the landscape scale to maintain spatially heterogeneous patches of different fire history (Kelly *et al.* 2012). According to Barrett *et al.* (2013) an understanding of the patterns of the spatially heterogeneity is important for predicting future fire regime shifts at regional scales. Within this context any inappropriate fire regimes would negatively impact organisms within the fire prone ecosystem at the landscape scale (Avitabile *et al.* 2013) and act as a key threatening process to some of the ecosystems organisms (Avitabile *et al.* 2013).

Wildfire regimes have changed over time for any given climate or vegetation group due to human influence (Martin and Sapsis1991) and are expected to change (Thonicke *et al.* 2013) unexpectedly and rapidly in response to future climate change. This change will impact on all fire prone ecosystems on the planet changing the complex spatiotemporal patterns of vegetation biomass, composition, and structure (McKenzie *et al.* 2011).

2.4 Australian fire regimes

Fire is an essential process in many Australian landscape and its ecosystems, playing a vital role in plant evolution, distribution, and adaptation (Watson *et al.* 2004). Cary *et al.* (2012) state that fire has been evident in Australia from charcoal records since the Paleogene Period (65 mya) and throughout the Neogene period to present. According to Lynch *et al.* (2007) fires have been a consistent feature within the Australian landscape, increasing in importance with the evolution of the sclerophyll vegetation and in determining the distribution of fire sensitive vegetation such as rain forest communities (Bowman 1998;

Enright and Thomas 2008). Natural fire regimes created self-sustaining eucalypt ecosystems, an important evolutionary driver of the Australian landscape, creating a dynamic equilibrium (Jurskis 2005) maintaining eucalypts and other sclerophyll taxa as the dominate species. It is also suggested that fires may increase within periods of climate warming, which is supported by the close relationship between fire and ENSO (Lynch *et al.* 2007; Cary *et al.* 2012). Mooney *et al.* (2011) suggest that the important evolutionary role of fire in Australia led to the development of plant responses to fire with morphological and reproductive adaptations to fire such as resprouting and obligate seeders. The palaeoenvironmental records show a consistent response to climate change by fire and that this response is rapid, where there are fewer fires under colder climates compared with warmer climate with a greater number of fires (Mooney *et al.* 2012).

Enright and Thomas (2008) suggest that almost all fire ignitions prior to the arrival of people on the Australian continent would have been the result of lightning strikes and that the frequency of ground flash lightning shows a strong north to south gradient. This is in agreement with Murphy *et al.* (2013) in that Australian fire regimes are related to the latitudinal gradient in the summer monsoon, where frequent low intensity fires occur in the tropical rainfall regions of the north with infrequent high intensity fires in the temperate southern regions (Murphy *et al.* 2013). The latitudinal gradient is postulated as the main driver of fire regime on the continental scale by Murphy *et al.* (2013) and not biomass accumulation. Further Murphy *et al.* (2013) states that Australian fire regimes are relatively frequent surface fires of grasses and herbage in the different vegetation types of the continent. The spatial arrangement of fire regimes and the influence on the viability of species is poorly understood in an Australian context (Driscoll *et al.* 2010).

Collins *et al.* (2013) found that precipitation regimes play an important role within wildfire regimes including spatial fire severity. Future fire management strategies will need to be adaptable to spatial variation in mean annual precipitation with changes in future climates. In support of Driscoll *et al.* (2010), Haberle *et al.* (2010) states that “current severe drought and plant mortalities are increasing fire hazard and raising concerns about the trajectory of post-fire vegetation change and future fire regimes” (p. 80) in Australia. Indicating further research is required to answer what possible changes to vegetation may arise and what are possible future fire regimes.

Bowman *et al.* (2012) says that there is mounting concern that wildfire frequency and intensity could increase with climate change. Fire regimes are affected by fire weather and fuel characteristics, and there is evidence of an increase in fire weather in southern Australia. Dunlop and Brown (2008) question as to how we should respond to the changing fire regimes. They suggest that we allow change, however manage the consequences of a changing fire regime, ensuring habitat survival for sensitive species and human infrastructure.

Fire regime at the landscape scale level involves a range of spatiotemporal disturbance variables which has led to the belief that diversity in fire regime would promote biodiversity (Faivre *et al.* 2011). According to Pastro *et al.* (2011) there is evidence that ecosystem heterogeneity created by burning patches of vegetation can increase biodiversity, an example is the spinifex grasslands of central Australia where fire heterogeneity increases mammal and reptile diversity. However, Robinson *et al*. (2014) found that fire history and severity in mixed eucalypt forest of south-east Australia played an important role in avian refuges. Patches not burnt for a long time (greater than 20 years) supported 20 to 40% more species with 56% more individuals compared to patches of short time-since-fire (less than 3 years) where bird abundance was found to be approximately 20% lower. Further Malleefowl (*Leipoa ocellata*) prefer habitat with fire return intervals of not less than 40 to 60 years and are threatened by extensive homogeneous frequent fires in both west and east Australia (Parsons and Gosper 2011). The ecological balance in many Australian eucalypt ecosystems has been upset through changes in fire regimes with the exclusion of fire by Europeans, which has resulted in a decline in both the abundance and health of these systems (Attiwill 1994; Jurskis 2005a; Jurskis 2005b).

2.5 The Great Sandy Region

In Queensland fire has been recorded from charcoal taken from sediment core samples as early as 130 000 years ago as a result of dry conditions at the end of the penultimate glacial period in combination with the phase of high ENSO activity (Moss and Kershaw 2007). Evidence of anthropogenic fires first date to around 45 000 years ago with the advent of Aboriginal occupation and fire stick practises (Fensham 1997; Turney *et al.* 2001; Moss and Kershaw 2007; Moss 2008). Aboriginal burning continued until the settlement of Europeans in the 1800s when fire regimes were disrupted dramatically with an increase in fires (Haberle *et al.* 2010) as supported by an increase in the charcoal accumulation rate during that period. According to Whitlock and Larsen (2002) charcoal peaks in the stratigraphic record would have been deposited during and after the fire, making it difficult to infer levels of fire intensity or fire size in the past based on charcoal abundance and that charcoal peaks reveal more about the taphonomic history of charcoal within the lake and watershed after fire (Whitlock & Millspaugh 1996, p. 14). Further Higuera *et al*. (2005) did not find any relationship between the magnitude of charcoal peaks and fire intensity.

The GSR is dominated by eucalyptus woodlands, mangrove forests, peat swamps and coastal heath plant communities (Figure 2.2) that are fire promoting and highly flammable (Moss *et al.* 2012) and rainforest. Fraser Island has important mycorrhizal fungi occurring naturally in the sand providing nutrients for the diverse plant community occurring on the island (Kurtböke *et al.* 2007). In the GSR fire history records have been collected and are well documented using radio-carbon dating and charcoal records. Moss *et al.* (2013) states that there are “three records of vegetation and burning from North Stradbroke Island provide the first continuous records extending past the Last Glacial Maxima” (p. 270) to around 47,000 years ago. According to Moss *et al.* (2012) research undertaken at Myora Springs on North Stradbroke Island indicates a link between a warm ENSO and drought between 800 to 500 years ago that promoted an increase in burning and flammable vegetation such as open eucalyptus forests and Melaleuca paperbark swamp (Moss *et al.* 2011).

Research undertaken by Longmore (1997) shows that the GSR, and in particular Fraser Island, has undergone substantial changes in vegetation community structure over time from climax through to advanced retrogressive stages. The vegetation has changed from a predominantly rain forest with *Araucaria* spp. to a dryer rainforest with *Podocarpus* spp. between 600ka to 350 ka to more sclerophyllous forest until before the Last Glacial Maximum (LGM). This change in the plant community was primarily driven by climate change and accelerated by increased burning. According to Longmore and Heijnis (1999) evidence of fire is low at the start of the record, but increases from around 350 ka to the LGM, thereafter increasing with frequent fire records during the Holocene. Changes to the island’s vegetation are thought to have been through succession, fire and climatic change (Longmore and Heijnis 1999). Around 500 years ago a cool ENSO persisted and the vegetation shifted once again to a fire sensitive Casuarinaceae dominated landscape

(Moss *et al.* 2011). According to Moss *et al.* (2012), there was a regional shift in fire regimes of the Sand Masses associated with European settlement, with a decline in burning and an expansion in eucalyptus and corresponding decline in Casuarinaceae, which needs further investigation as to why there was a decline in fire sensitive plants (Moss personnel communication).

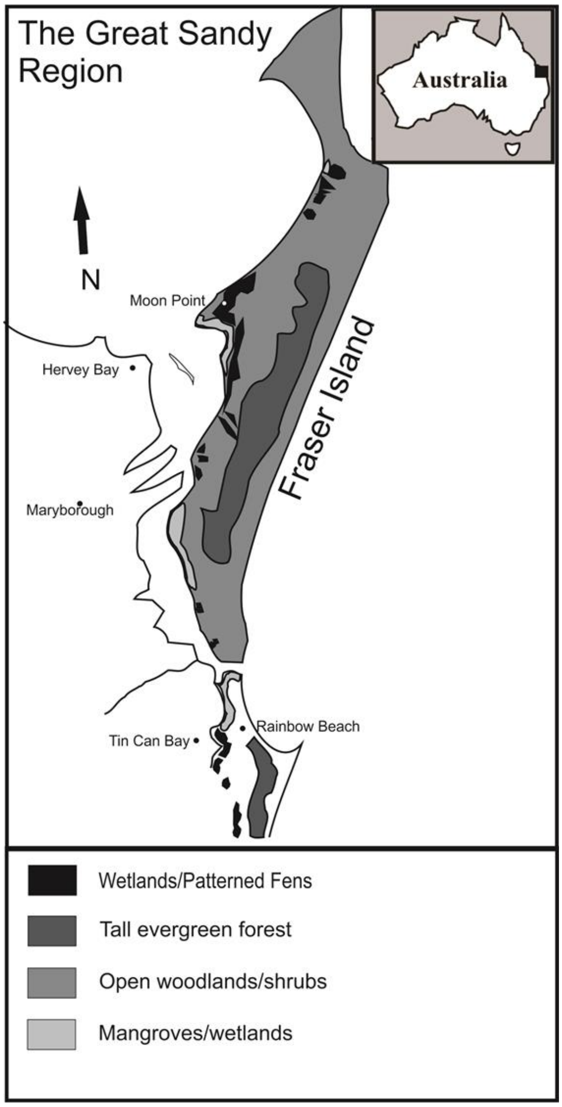


Figure 2.2. Location of the Great Sandy Region in relation to the Australian east coast with four dominant vegetation communities of the region.

Late Quaternary fire regimes of the Sand Masses of south east Queensland are extremely complex according to Moss, *et al.* (2012a) site location could be a factor in determining the dominance of either anthropogenic or climate factors as the control (Moss *et al.* 2012a). Questions resulting from this research asked by Moss *et al.* (2012a) include what are the long-term ecological impacts of altered fire regimes on biodiversity, what are the vegetation shifts across the region and what factors are behind the higher fire regimes of the pre-European arrival to the region?

According to Jurskis (2005) fires were ignited by lightning and Aborigines over time and space however the extent of these fires were governed by climatic and edaphic conditions. Research by Fensham (1997) shows that Aboriginal burning during the initial contact with Europeans was prevalent along the coast of Queensland with high frequency of fires occurring compared with infrequent burning undertaken inland and with limited fires being lit in the summer and spring months.

Present fire regimes of the region are managed by the Queensland Government's Department of National Parks, Recreation, Sport and Racing through prescribed burning and wildfire management. Fire is managed according to fire protection policy, biodiversity conservation and through fuel reduction activities. According to Hocking (1998) monitoring the impacts of fire on Fraser Island vegetation consists of two elements, fixed point photography of before and after (fire events) and fixed plots using the standard Braun-Blanquet method. Prescribed burning is undertaken for ecological burning, hazard reduction burning, wildfire suppression activities, or burning for weed control and site rehabilitation for the purpose of the protection of life and property, and to ensure the maintenance of biodiversity through continued natural processes (Queensland Government 1994).

2.6 Conclusion

Fire has been recorded since the advent of the first vascular plants, and fire regimes have shifted with changing climatic and atmospheric oxygen levels (Pausas and Keeley. 2009) driving evolutionary processes of ecosystems such as distribution and composition of species. These shifts in fire regime continued and were modified by humans through migration, settlements, and land clearing practices over the millennia further impacting ecosystem dynamics and composition. European settlers in Australia have had a substantial impact on fire regimes altering natural ecosystems to and from fire tolerant species depending on the fire regimes at that time.

According to Driscoll *et al.* (2010) quantification of fire regimes is still unknown and in its infancy and how species are influenced by the spatial arrangement and temporal sequences of fires. Parr and Andersen (2005) suggest that the "ecological significance of different burning patterns is still unknown and has led to fire-management plans being based on pyrodiversity rhetoric that lacks substance in terms of operational guidelines and capacity for meaningful evaluation" (p. 1610). However prescribed burning has been carried out in Australia since the 1960s to manage and reduce fuel loads in southwestern eucalyptus forests and continues to be an important management tool for reducing impacts of wildfires (Burrows and McCaw 2013). Further according to Wardell-Johnson *et al* (2006) burning fine-grained mosaics of vegetation at different post-fire stages is a management strategy used to maintain habitat heterogeneity across space and time as there is evidence that diverse fire regimes benefits biodiversity in fire-prone environments.

Further unknown factors are how herbivory, predation, fragmentation, invasive species and weather interact with fire to alter species responses to fire directly, or via changes to the fire regime (Driscoll *et al.* 2010). Lastly what are the long-term ecological impacts of an altered fire regime for the conservation of south east Queensland's biodiversity (Moss *et al.* 2012b). The answers to these questions are paramount to successful management of future fire regimes of the region. However, it is not possible wait until these issues are addressed, as that will lead to undesirable consequences. Prescribed burning is needed and should be guided by research, in a dynamically integrated approach, which protects human society, ensures conservation of biodiversity and maintains ecological structures and functions. The interval between prescribed fires needs to be sufficient to reduce fuel loads for hazard reduction and result in ecologically acceptable outcomes (Burrows *et al*. 2008).

Chapter 3. Fire Regimes and Invasive Species: A Changing Landscape

Abstract

Tropical and semi-arid savannas are extremely fire prone (Lawes 2011) with dry season fires being a feature of this landscape in Australia. Historically fire has been an integral part of shaping the landscape into its present state. Initially occurring naturally due to lightening fires, followed by Aboriginal fire stick farming practices (Burrows and Christensen 1990; Bowman *et al*. 2004; Pausas and Keeley. 2009; Russell-Smith *et al*. 2013; Jurskis 2015), which has had a substantial impact on the landscape. With the arrival of the Europeans from the late 18th century onwards fires were perceived negatively, which resulted in there prevention and exclusion from the Australian landscape (Baker *et al.* 2015; Beringer *et al.* 2015). This resulted in the increase of fuels and a change in fire regimes, with larger, higher intensity uncontrollable wildfires able to occur, which had a negative impact on native vegetation (Jurskis and Underwood 2013). Today fire is seen as an important and integral part of the natural ecosystem of tropical and semi-arid savannas and is used as a land management tool where possible (Walsh *et al.* 2014). However pastoralists involved in the cattle industry require improved pastures through introduced grasses and this has resulted in a number of exotic grasses invading native tropical and semi-arid ecosystems (Driscoll *et al.* 2014). Five of Australia's worst environmental weeds are introduced pastoral grasses, which received little attention until 1991 (Low 1997). Changes to fire regimes is a consequence of the spread of tropical Africa grasses with increased fuel loads and fire intensity (Wilson and Mudita 1999), which were deliberately introduced by pastoralists (Woinarski 2005). This invasion has altered the dynamics of the fire regimes through higher fuel loads, resulting in higher intensity, more severe fires and a change in vegetation response to fire (Setterfield *et al.* 2013; Grice *et al*. 2013).

3.1 Introduction

Fires are a major ecological driver (Bowman *et al*. 2003*;* Bond and Keeley 2005; Jurskis 2005; Bowman *et al.* 2012; Stewart and Moss 2015; Keeley *et al.* 2016) of tropical and semi-arid savannas that occur in monsoonal climates. This is related to the latitudinal gradient in summer monsoon activity (Stewart and Moss 2015) making these one of the most fire-prone landscapes in the world with large areas experiencing annual or biennial fires (Vigilante *et al.* 2004; Russell-Smith *et al*. 2013; Murphy *et al*. 2015; Russell-Smith *et al*. 2017). Fires are mostly constrained to the dry season months where fire intensity is largely determined by climatic conditions (i.e. low precipitation) and fuel loads (Boer *et al.* 2016; Prior *et al.* 2016). However, humans have altered fire regimes by artificially setting fires that greatly extends the burning season and increasing ignition events that are well above the natural patterns linked to lightning strikes (Archibald *et al.* 2012).

Evidence of anthropogenic fires date back to approximately 45 – 50 ka (Bird *et al.* 2008; Moss *et al.* 2013; Jurskis 2015; Moss *et al.* 2016) with the advent of Aboriginal occupation and agricultural fire stick practices (Singh *et* al. 1981; Burrows and Christensen 1990; Fensham 1997; Turney *et al.* 2001; Bowman et al. 2004; Moss and Kershaw 2007; Moss 2008; Pausas and Keeley. 2009; Russell-Smith et al. 2013 Stewart and Moss 2015; Jurskis 2015). According to Ward *et al.* (2001), Aboriginal burning records based on the analysis of fire-scars from grasstrees (*Xanthorrhoea*) show that burning was undertaken every 3 to 5 years (Horton 1982), however with the arrival of Europeans there was a decline in fire frequencies with an increase in fire intensity and severity (Jurskis *et al*. 2003) associated with logging, although the fire frequencies increased initially (Haberle *et al.* 2010), which is supported by an increase in charcoal deposits around that time (Mooney *et al.* 2011; Stewart and Moss 2015). In the early 20th century fire exclusion and suppression was introduced as a measure to prevent uncontrolled wildfire, which was followed by prescribed burning between 1950 and 1960, both of which were not successful in achieving wildfire prevention goals (Ward *et al.* 2001; Jurskis *et al.* 2003), but rather resulting in less frequent and more intense fire activity. There is also evidence that with these changes in fire regimes there may be vegetation thickening (i.e. increased arboreal vegetation) occurring, which may contribute to a further increase in fuel loads (Murphy *et al.* 2014).

The introduction of perennial grasses to improve grazing in tropical and semi-arid regions of northern Australia has resulted in some exotic grass species such as Gamba grass (*Andropogon gayanus*), Mission Grass (*Cenchrus polystachios* syn. *Pennisetum polystachion*) and Guinea grass (*Megathyrsus maximus:* Synonyms *Panicum maximum*

Jacq. var. trichoglume) becoming pests and invasive (Lonsdale 1994; Driscoll *et al.* 2014; Douglas *et al.* 2016). The invasion by these exotic grasses has serious implications for ecosystem function (Strauss *et al.* 2006; Sands and Goolsby 2013;Turpie 2016) especially for Fraser Island and the Great Sandy Region where for example Guinea grass has become established, as these grasses alter fire regime dynamics through increasing the distribution and abundance of fine fuels. With increased fine fuels there is a serious danger that there will be an increase in fire frequency and intensity resulting in higher severity burns and higher vegetation mortality, with possible local species extinctions and habitat modification or change (Enright 2011; Kukavskaya *et al.* 2016). This paper will provide a review of the natural historical fire regimes for northern and eastern Australia and examine the impacts of invasive exotic grasses on fire frequency and intensity. Natural fire regimes are described in terms of frequency, season intensity and spatial variability (Jurskis *et al*. 2003) which includes human occupation and maintenance of traditional fire management practices (Yibarbuk *et al.* 2001).

3.2 Study area

Introduced perennial grasses are a major issue in terms of impacting fire regimes in tropical and subtropical Australia (Figure 3.1). The climate of the region is governed by the seasonal movement of the Intertropical Convergence Zone (ITCZ), which migrates southward over northern Australia during the summer months of November to March, bringing with it the Australian summer monsoon rain and north-westerly winds (Reeves *et al.* 2013). North-eastern Australia's wet season is extended by the additional moisture received from the humid easterlies of the Coral Sea and equatorial Pacific; however the Great Dividing Range acts as a barrier to easterly rain, creating the inland tropical savannah. North-western Australian rainfall is influenced by the local sea-surface temperatures of the Indian Ocean and Timor Sea (Reeves *et al.* 2013). Subtropical Queensland experiences warm summers and cooler winters with rainfall influenced by the latitudinal gradient in the summer monsoon. During the winter months of June to September the ITCZ migrates back to the northern hemisphere providing conditions for a warm but dry winter over northern Australia and a cool dry south east Queensland (Reeves *et al.* 2013; Gensac *et al.* 2016; BOM 2017).

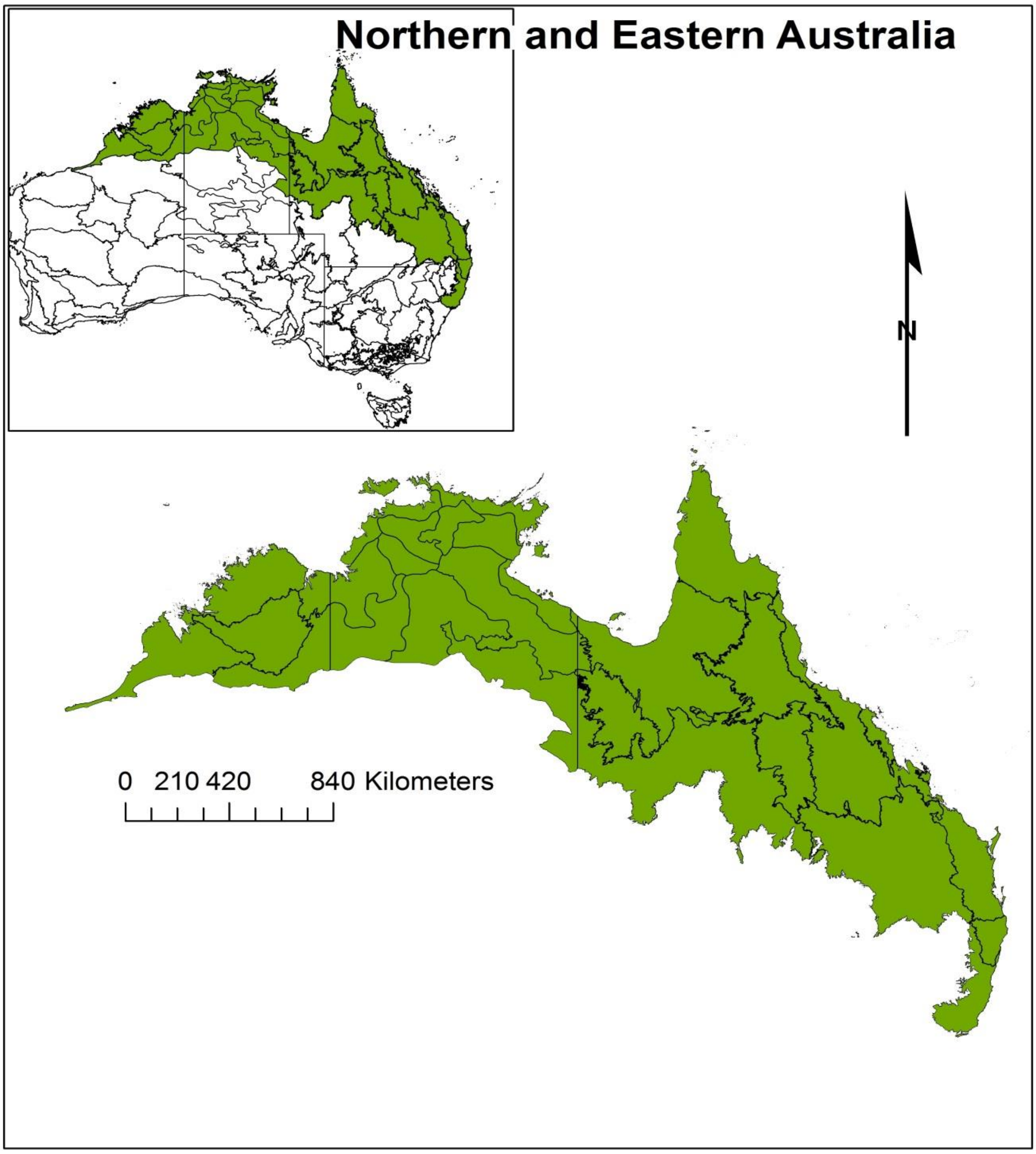


Figure 3.1. Map of northern Australia showing region of invasion by perennial grasses.

The common soils of the region are dystrophic and petroferric Red Kandosol and petroferric, eutrophic and dystrophic Brown Kandosol, all commonly known as Yellow Earths (McKenzie *et al.* 2004). Generally Kandosols are found on extensive, level to gently undulating plains and on mesas of varying extent. Kandosols soils are well drained permeable soils that range from near-coastal high rainfall regions to the arid and semi-arid interior (McKenzie *et al.* 2004). Vegetation of the region is varied, but is generally dominated by open forest and woodlands (primarily eucalypt and acacia trees), with an understory of native grasslands.

3.3 Discussion

Perceptions of fire severity are influenced by features related to intensity, such as flame height, rate of spread, extent of the fire front and magnitude of the smoke pall (Whelan 1995). It is now known that one fire is not like another with respect to its impacts on the biota according to Schwartz *et al.'s* (2016) research on the Great Smokey Mountains. A fire may have a negligible effect on the community within which it occurred; whereas another fire that is less severe according to human perceptions (Whelan 1995) can alter the community structure and species substantially. Therefore time between consecutive fires or fire frequency can have a marked impact on vegetation, independent of the fire intensity. Other features of fire may also be important in determining the effects of fire on the biota, such as the timing between fires (known as frequency), season of burning, the extent (the patchiness), and the sequence of fire. The sequence of fire intensity may play an important role in patch burn mosaics and constituent specie/s extinction on a local or regional temporal and spatial scale. In brief, what are the impacts of a hot fire, followed by a hot fire, then a cool fire, then a moderately intense fire and another hot fire? The main regulating factor for a fire regime is climate. Past climate influences fire by determining the characteristics and distribution of plant communities and current climate determines natural ignition sources and subsequent fire behaviour (Whitlock *et al.* 2003; Bowman *et al.* 2009). Therefore to have an understanding of the Australian landscape as it is today we need to look at both present and past fire regimes.

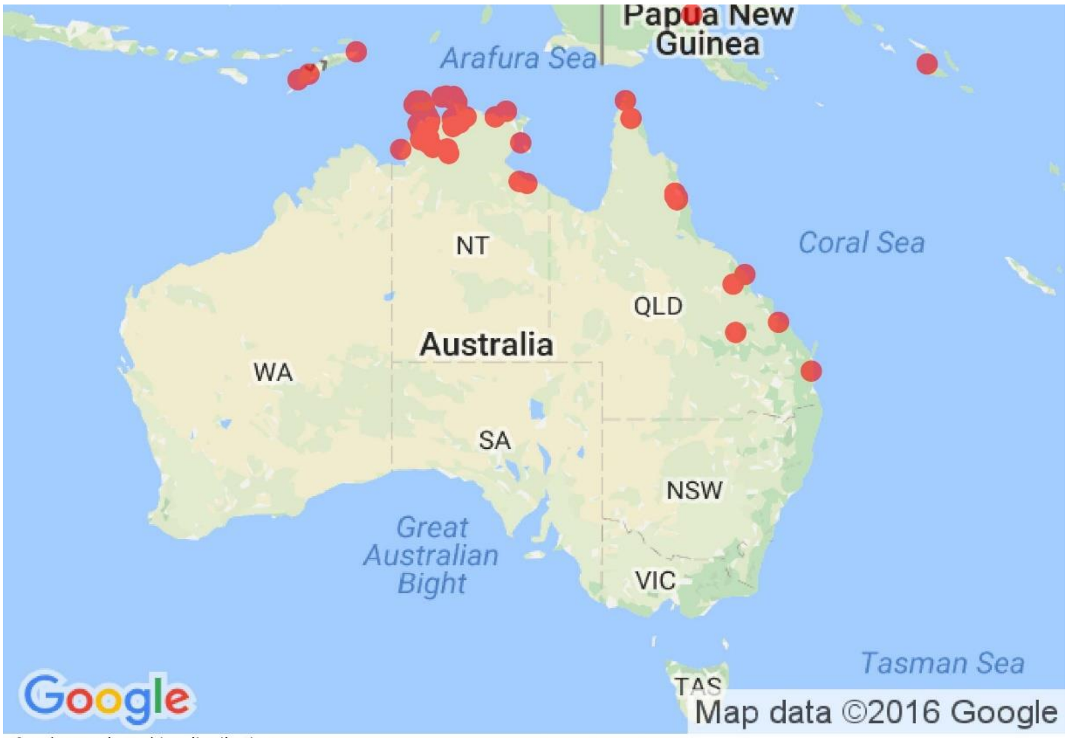
Tropical, semi-arid savannas and subtropical grasses are a mix of annual and perennial that rely on the wet season and summer rainfall as a growth trigger (Keya 1997). For instance, native annual *Sorghum* grasses are opportunistic in that they increase and spread due to an increase in fire frequency (Andersen *et al.* 2005; Wang and Niu 2016). Grasses of the region include spinifex and hummock grasses, Mitchell grass, mixed tussock and tall grasses. The various grasslands of savannas and the subtropics are dominated by different species depending on soil type and rainfall (Tropical Savannas CRC 2016). With high wet season rainfalls, there is a corresponding high annual grass production and *vice versa* if there is low wet season rainfall (below the average of 1600mm), grass biomass (fuel loads) is determined by the rainfall of each wet season. However on average the annual biomass of *Sorghum* grasses (depending on soil type and topography) varies from half a ton per hectare (Oven Dry Weight [ODW]) up to four tons per hectare (Figure 3.2). The dynamics of the fuel, its arrangement, distribution and density play a role in the fire intensity and rate of spread.

Exotic plant invasions are one of the world’s worst ecological problems (Vitousek *et al.* 1996; Richardson and Allsopp 2000; Rossiter *et al.* 2002). All successful plant invaders will change the structure, composition and habitat quality of native plant communities (Rossiter *et al.* 2002; Grice *et* al. 2013; Wagner and Fraterrigo 2015). Weeds affect ecosystems they invade in a number of ways with often obvious dramatic changes. For example weeds sequester resources that would otherwise be available to native plants (Alldred *et al.* 2016). Further the pathways and rates of water and nutrient cycling are altered compared to non-invaded sites (Williams and Baruch 2000). Invasive plants also alter the structure of the vegetation they invade and alter the ecosystems fire regime (Grice 2004; Bajwa *et al.* 2016). This occurs when invasive plants influence fuel loads, fuel distribution, fuel continuity or the timing of fuel curing (drying of the fuels), with repercussions for the intensity, timing, frequency and sequence of fires (Black *et al.* 2016).

Figure 3.2**.** Annual native grass species showing distribution and arrangement. Annual *Sorghum intrans* grass with a biomass of 2.45 ton per hectare (Oven Dry Weight ODW) showing the typical arrangement and distribution patterning of this grass in a tropical Savanna, with a height of 1.1metres as seen in the early dry season (May). Photograph: Philip Stewart**.**

Invasive grass species can cause dramatic changes in fire regimes through a positive feedback cycle, called a “grass-fire cycle” (Pausas and Keeley 2014b; Wagner and Fraterrigo 2015). The grass-fire cycle occurs when alien grass invades an area and increases the abundance of fine fuels, which increases fire frequency and intensity (D'Antonio and Vitousek 1992). This can result in a decrease in native tree and shrub cover and abundance, which further facilitates more grass invasion, which in turn increases the risk of future higher intensity and frequency of wildfires in an ever increasing self-perpetuating fire cycle (Flory *et al.* 2015).

Mission grass (*Cenchrus polystachios* syn. *Pennisetum polystachion*) from West Africa was introduced into northern Australia between 1940 to 1950, becoming established in the Northern Territory in the early 1970’s then spreading across the Northern Territory to Queensland (Figure 3.3) where it is now naturalized (Australian Commonwealth Government 2014). According to Douglas *et al.* (2004) very little is known about the potential impacts of Mission grass in savanna habitats. However Douglas *et al.* (2004) found that Mission grass fuel loads were approximately five times greater than that of native grasses. The greater fuel loads as a result of Mission grass invasion could result in higher intensity fires (Douglas *et al.* 2004; Setterfield *et al.* 2013), and a fire driven transition from forest to grassland (Skarpe 1992; D'Antonio and Vitousek 1992; Rossiter *et al.* 2003; Bond, 2008). Mission grass poses a serious risk to Fraser Island and Great sandy Region if it becomes established, as it has been identified south of the region (Figure 3.3).



Cenchrus polystachios distribution

Figure 3.3**.** Map of present distribution of Mission Grass. Source: Australia's Virtual Herbarium.
http://avh.ala.org.au/occurrences/search?taxa=Cenchrus+polystachios+#tab_mapView

Gamba grass (*Andropogon gayanus*) was introduced into the Northern Territory as a pasture grass in 1931 and is now well established outside pastoral systems in the Northern Territory, Western Australia and Queensland (Figure 3.4) where it was introduced in 1942 (Australian Commonwealth Government 2014). Concern has been focused on the increased fuel loads of Gamba grass and possible intense late dry season fires resulting in a grass-fire cycle in tropical and semi-arid savannas (Setterfield *et al.* 2010; van Klinken *et al*. 2013).

Gamba grass invasion can cause a substantial change to the fire regime of tropical and semi-arid savannas. It is expected that due to this species high rate of production, it will result in an increase of high intensity fires up to eight times more intense than native grass species even in the early dry season. Further, fire frequency is expected to increase as Gamba grass can support more fires than annually due to the fact that Gamba grass fuel loads may be up to 24 ton per hectare and post-fire fine fuel loads up to 6 ton per hectare have been recorded left standing, which will also carry fire (Allan 2005. pers comm., 2nd April).

The formula for calculating fire-line intensity is $I = Hwr$ (known as ***Byram's Fire Intensity Equation***).

Where: I is fire-line intensity (kW/m)

H is heat yield of the fuel (kJ/kg)

W is fuel load consumed (kg/m^2)

r is the rate of fire spread (m/second)

If we assume for convenience that the heat yield is a *constant* @ 18 000 kJ/kg, the formula

is:

$$\boldsymbol{I = \frac{wr}{2}}$$

Where: ***w*** is measured in t/ha and

r is measured in km/h

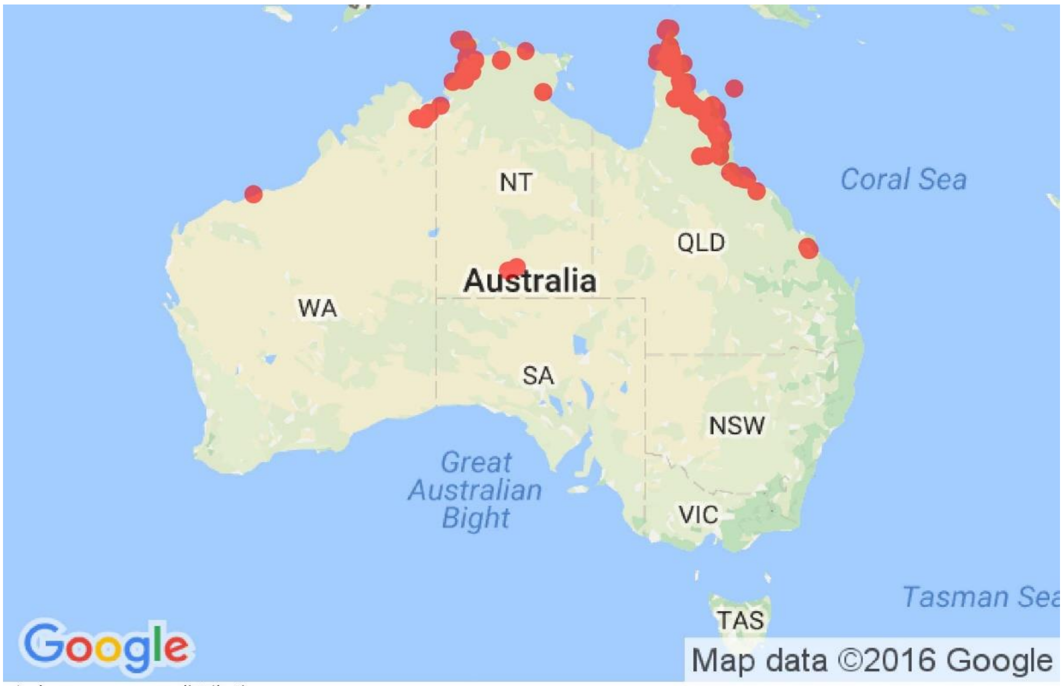


Figure 3.4. Distribution map of present extent of Gamba grass invasion across northern Australia. Source: Australia's Virtual Herbarium http://avh.ala.org.au/occurrences/search?taxa=Cenchrus+polystachios+#tab_mapView

Guinea grass (*Megathyrsus maximus*) also known as green panic grass is a large perennial grass native to Africa, Yemen and Palestine that was introduced to Queensland (Figure 3.5) by the Acclimatisation Society to 22 locations between 1865 and 1869 (Eyles *et al*. 1985; Clements and Henzell 2010; van Klinken *et al*. 2015). Guinea grass is not a prohibited or restricted invasive plant under the Queensland Biosecurity Act 2014 and it is used extensively as a fodder grass (State of Queensland 2016). Guinea grass is promoted by fire, re-shooting and having abundant seed germination. Further fire intensity is increased by increased biomass (Douglas *et* al. 2016) of fine fuels causing the death of trees that survive fires fuelled by a natural understorey (Williams 2008; Pettit *et al*. 2016). Grice *et al*. (2010) found that Guinea grass in Kakadu National Park produced much higher above-ground biomass than local native grasses, fuelling destructive high intensity fires.

van Klinken *et al*. (2013) states that 155 species of grass are recognised as being naturalized in Australia with only 21 being identified as high impact and of these only four are considered high-impact for two sectors, the environment and pastoral industry, Guinea grass being one of these four grasses. It is a highly successful invader in the tropics and temperate regions spreading by seed and is fire resistant, allowing it to spread quickly into gaps left by native vegetation after fire (CABI 2017). Guinea grass has been found to dominate forest understory where it produces high biomass that promotes high intensity fires and has caused marginal attrition of rainforests through repetitive fires (CABI 2017). According to Sands and Goolsby (2013) more than ten years of extreme weather events in eastern Australia have influenced the growth of invasive African grasses such as Guinea grass increasing its ability to invade natural ecosystems and to build up fuel loads during droughts and dry periods and heavy rainfall events, increasing fire intensity. As figure 3.5 shows, Guinea grass has become established across eastern Australia including the Great Sandy Region (State of Queensland 2013) and within and around all townships on Fraser Island in south eastern Queensland (L. Behrendorff 2017, pers comm., 16th June).

Figure 3.5. Present distribution and extent of Guinea grass (*Megathyrsus maximus*) in Australia. Source: Australia's Virtual Herbarium http://avh.ala.org.au/occurrences/search?taxa=Panicum+maximum+var.+trichoglume#tab_mapView

As stated above, African tropical grass invasion results in an increase in fine-fuel loads, which directly affects the fire regime by increasing fire frequencies and intensities (Rossiter *et al.* 2002; Williams 2008), including the sequence of fires, such as a high intensity fire followed by a cooler fire will no longer occur as fuel loads will dictate a higher intensity burn. This may result in a “grass-fire cycle” with an associated reduction in shrub and tree species abundance as a result of mortality caused by intense fire (Figure 3.6 and 3.7).

Figure 3.6. Gamba grass fire showing torching of canopy in a tropical savanna. A high intensity Gamba grass fire with torching of tree canopy cover resulting in high shrub and tree mortality opening up the savanna to a “grass-fire cycle” (Photograph Courtesy of Bushfires Council NT)

Studies have shown that Gamba, Mission and Guinea grasses are likely to have impacts on three key determinants ecosystem functioning, fire regimes of which we are interested, nutrient availability and water availability. Further, these grasses have the potential to alter not only the amount of resources available, but also the seasonal timing of these resources (Rossiter *et al.* 2004; Setterfield *et al.* 2014).

These changes are likely to have serious consequences for the structure and functioning of ecosystems. With the alteration in fire regime due to increased fuel loads, changes in fire frequency and intensity (Rossiter *et al.* 2002; Woinarski 2005; Williams 2008; CABI 2017), long-term outcomes are high mortality of tree species (Figure 3.7), with local species extinctions (especially obligate seeders), with an extensive reduction in canopy cover and live basal area, and a large reduction in the abundance of deciduous taxa

(Williams *et al.* 1999). Other impacts appear to be a reduction in flowering and fruiting in the dominant eucalypts and non-eucalypts.

Apart from the obvious high costs associated with alien grass eradication programs, invasion by these African grasses have other more serious costs with respect to changes in biodiversity, community structures and local species extinctions. Andersen et al. (2005) have shown that tree mortality will increase with increasing fire intensity. Studies to date on tropical savanna show that fire regimes have complex effects on savanna tree dynamics, influencing mortality and recruitment in different ways (Levick *et al.* 2009). A marked decrease in diversity and floristic patterns has been noted when ecosystems are invaded by these grasses due to high-intensity fires (Rossiter *et al.* 2003; Levick *et al.* 2015; Wang and Niu 2016). Recruitment is reduced due to high seed mortality of sorghum species impacted by high intensity fires that result from invasive grass fires.

Figure 3.7. High tree mortality post Gamba grass fire showing some resprouting from lignotubers of eucalypts. A dense stand of Gamba grass can be seen in the background, almost wall-like in appearance, showing extent of invasion by this grass specie in a tropical Savanna. (Photograph: Courtesy of Bushfires Council NT).

Impacts of high intensity fires caused by the increase in fuels from invasive grasses on faunal populations are poorly understood and require more research. However studies on tropical savanna fires have shown negative impacts on frog, lizard, and bird species (Andersen *et al.* 2005; Ledo and Colli. 2016; Greenberg *et al.* 2016). The physical changes to the habitat by erosion, hydrological pattern change and other abiotic factors (topography) will greatly influence the habitat's response to invasion by these grasses and has received no attention to date and the author feels this is critical in the understanding on habitat response and dynamics to invasion and habitat resilience.

3.4 Conclusion

Invasion by invasive grasses is likely to result in substantial changes to fire regimes. In tropical and subtropical ecosystems in the early dry season, the large fuel loads will support greater intensity fires of up to eight times more intense than native grass fuelled fires (Rositter *et al.* 2003;Williams 2008; Moon and Adams 2016; Fill *et al.* 2016). Further fire frequency is expected to increase due to invasion by these grasses as they have the potential to support fire more than once a year. In native ecosystems seasonal burns are patchy and limited in extent due to the discontinuous nature of the fuel loads and fire weather (Whelan 1995; Weier *et al.* 2017). However Guinea, Gamba and Mission grasses are expected to increase the continuous fuel load substantially, which in turn will increase the probability of uniform fires over extensive areas. These high intensity, uniform and frequent fires will drastically alter the vegetation structure and composition of communities. High intensity uniform fires in savannas have shown to reduce savanna woody species recruitment (as stated above) and cause considerable tree mortality (Williams *et al.* 1998; Smit *et al.* 2016).

Studies have shown that invasion by Guinea, Gamba and Mission grasses and their effect on fire intensity in Australia is similar to that resulting from the invasion of African grasses in other ecosystems (Rossiter *et al*. 2003), and in three cases the landscape has been grossly modified by this process. These fire promoting grasses can be described as ecosystem transformers that have the potential to alter the community structure (Rossiter-Rachor *et al.* 2016) and the nutrient, water and carbon cycling processes over large areas of tropical savannas. Of these three species, it can be seen that Guinea grass has become widely naturalized across eastern Australia and has the greatest potential threat to the GSR. Gamba grass has also become established within the vicinity of the GSR and may develop in a potential threat. Mission grass is mainly restricted to northern Australia but its impacts on tropical vegetation provide insight into the potential impacts on fire regimes that exotic grasses may have in the GSR. Therefore, greater attention needs to be paid by all land managers on the potential impacts exotic grasses may pose to the natural ecosystems across tropical and subtropical Australia, particularly the potential threat, in combination with climate change and the altered fire regimes of the European settlement period (see Chapter 2), of Guinea grass becoming established within the iconic GSR, .

Chapter 4. Late Quaternary fire history and vegetation dynamics for Fraser Island, subtropical eastern Australia

Abstract

The aim of this study is to identify past and present fire regimes and changes in vegetation dynamics through the analysis of charcoal and pollen records taken from sediment cores from Fraser Island, subtropical eastern Australia. We undertook high resolution macro charcoal analysis using contiguous subsamples at 1 cm intervals of 1 cm^3 and pollen analysis using contiguous subsamples at 5 cm intervals of 1 cm^3 that were taken from the sediment cores collected at Moon Point, Fraser Island (Great Sandy Region National Park) to identify paleofire and vegetation histories for the area. The results showed that changing fire regimes and climate over the past 25,000 years within the study area have resulted in a change in predominately rainforest species (pyrophobic) to sclerophyll (pyrogenic) vegetation, with an increase in fire events during the drier mid-Holocene highstand period 7,000 – 6,000 years ago and a further marked increase in fire events around 200 years ago, which coincides with the arrival of Europeans. The analysis of macro charcoal (>125 microns), micro charcoal (>10 to <125 microns) and pollen indicate that Moon Point has been subjected to variable climate and fire regimes over the past 25,000 years, which has changed the dynamics of the vegetation from pyrophobic to pyrogenic and that climate has been an important driver in fire events that has resulted in this vegetation shift. Further with human settlement and a drying climate has seen an increase in fire events suggesting that the coexistence of anthropogenic fire regimes with the strong climate and fire relationships exist here.

4.1 Introduction

Fire has been shown to be a significant driver of ecosystem evolution, composition and distribution through its impact on biota (Bond and Keeley 2005; Pausas and Keeley 2009; Coneder *et al.* 2009; Bowman *et al.* 2011; Keeley *et al*. 2011; Batllori *et al.* 2013). Within Australia fire has long played a role in shaping the landscape, with increased fire frequency, associated with increased aridity, over the last five million years promoting the expansion of fire adapted sclerophyll vegetation across the continent (Hill 2004). According to Singh *et al.* (1981) the oldest continuous record of vegetation and fire history date back to approximately 350 ka years at Lake George, New south Wales. Currently, there is a debate into the relative roles of people and climate influencing fire regimes and vegetation since the arrival of Aboriginals at least 50,000 years ago (Hiscock 2013). Authors, such as Bowman (1998), Jurskis *et al.* (2003), Miller *et al*. (2005), Gammage (2011), Bird *et al*. (2013) and Jurskis (2015) argue that the Australian landscape should be viewed as an artefact of anthropogenic imposed fire regimes. In contrast, Horton (1982), Mooney *et al*. (2011), Notora *et al*. (2011) and Williams *et al*. (2015) suggest that climate plays the key role in understanding alterations in fire regimes and vegetation. However, both sides of the debate agree that European colonisation of Australia (beginning around 220 years) had the most profound impact on fire regimes and vegetation change over this time period, with fire initially used in land clearance for agriculture and other associated land-uses then being replaced by a policy of fire suppression, which has been a dominant fire management policy until relatively recently (Mooney *et al*., 2011; Adams 2013, Moss *et al*. 2016). Suggested environmental impacts of fire suppression have included vegetation thickening and the development of dangerous and destructive wildfires (Russell-Smith 2003; Adams 2013; Moss *et al*. 2015). Research by Burrows *et al*. (2006) found that with the departure of Aborigines from the western desert region of Western Australia and the cessation of their regular burning practices, the size and intensity of fires have increased substantially over the past 50 years. Changes in burning practices from Aboriginals to Europeans show similar outcomes for other Australian biomes (Burrows *et al*. 2006). Determination of the relative role of people and climate in influencing fire regimes and vegetation change plays a key role in improving fire management strategies through a better understanding of how fire can be used for hazard reduction, maintenance of ecosystem health and promotion of biodiversity.

To improve our understanding of the relative roles of people and climate in influencing fire regimes we undertook a multi-proxy investigation of a sediment core from a wetland in the World Heritage-listed Fraser Island, situated in subtropical eastern Australia (Figure 4.1). Fire is a natural occurring event on Fraser Island and is an important recycling mechanism to sustain plant growth (Donders *et al.* 2006). This island is dominated by a mix of fire adapted ecosystems, including heathlands, eucalypt forest/woodlands and wet sclerophyll forest, as well as pyrophobic communities such as rainforest and mangroves. The island is formed from a series of parabolic dunes deposited since the mid-Quaternary (~last 800,000 years ago) during periods of fluctuating sea-levels (Ward, 2006).

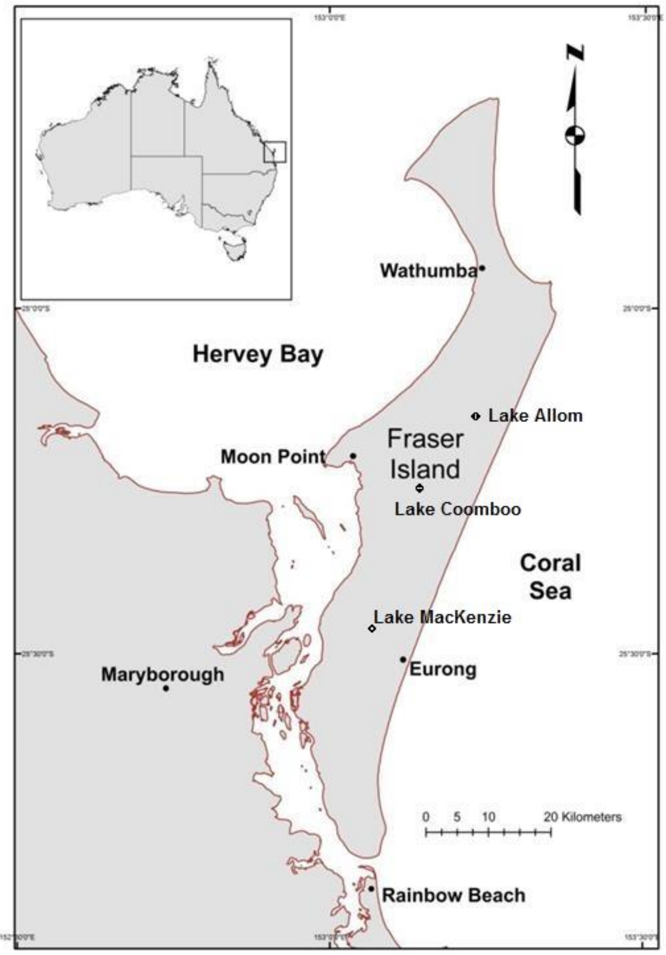


Figure 4.1. Map of Fraser Island and the Great Sandy Region showing the location of Moon Point on the west coast of Fraser Island. Insert map showing the location of Fraser Island in relation to Australia.

There are four paleoenvironmental sites on the Island that extend to and beyond the Last Glacial Maximum (LGM; ~20,000 years ago), three of these sites, Lake Allom, Old Lake Coomboo and Lake MacKenzie, are located in perched lake systems situated in the Pleistocene high dunes and all have a hiatus that coincides with the LGM (Longmore 1997; Longmore and Heijnis 1999; Donders *et al*. 2006; Hembrow *et al*. 2014; Atahan *et al.* 2015). The other record, which is located close to the study site discussed in this paper, was taken from a lowland *Empodisma minus* wetland and provides a 35,000 year record, which observes continuous sedimentation during the LGM (Moss *et al*. 2016). The aims of this study are to identify past fire regimes and vegetation history through the analysis of charcoal and pollen data, which will shed light on the relative roles that people and climate may have played in influencing fire frequency and intensity for the island, as well as examining if this had any impact on the islands unique landscapes, which will help with management of this iconic environment.

4.2 Study area

Fraser Island (K'Garri is the Indigenous Butchalla name) is World Heritage listed and one of five major barrier islands along the south-east coast of Queensland (Wardell-Johnson *et al.* 2015). It is the largest sand island in the world that has developed over the last 800,000 years (Peace 2001; Yizhaq *et al.* 2013; Gontz *et al.* 2015; Moss *et al.* 2015) forming part of the Great Sandy Region National Park that is situated off the east coast of Australia. The island is approximately 124 km long and 20 km wide with an average annual rainfall of between 1300 to 1700 mm/year with mean temperatures ranging between 14 °C in winter and 29 °C in summer (Figure 4.2). In summer the Inter-Tropical Convergence Zone (ITCZ) moves south pushing the subtropical ridge to 40^{o}S that exposes the island to the moist easterly trade winds producing the onshore easterly summer winds and in winter the ITCZ moves north allowing the subtropical ridge to move to 30^{o}S that brings the dry south-westerly winter winds to the island (Gontz *et al.* 2015).

Fraser Island was linked to the mainland and the Cooloola mainland sandmass for most of the last 800,000 years, apart from the higher sea-level interglacial periods, with both the island and Cooloola region consisting of extensive dune systems (Longmore 1997). There are more than 70 active transgressive dune fields (sand blows) present on Fraser Island (Figure 4.3), with 13 being greater than 100 ha and are mostly initiated as blowouts along the beach (Levin 2011). There are nine parabolic dune sequences that formed from the mid-Quaternary, including the oldest known dune sequence of approximately 730,000 to 800,000 years ago, and is still a key ongoing geological process for the island (Wardell-Johnson *et al.* 2015; Gontz *et al.* 2016). The dune minerals and nutrients are leached though the soil profile forming an iron-humus rich B-horizon that characterises the podosol soils (Thompson 1992), which are some of the world's deepest and most developed podosols with profile depths of the A1 and A2 horizons to the B-horizon as much as 30 metres in the oldest dunes (da Silva and Shulmeister 2016). The soils are composed of marine and aeolian sand deposits with few scattered bedrocks of basalt that have a significant geomorphic influence by interrupting longshore transport of sand and function to trap sand on the island (Johnson 2004). The central portion of Fraser Island is composed of complex Pleistocene high dunes that are vegetated, with numerous lakes occurring in the depressions between the dunes (Gontz *et al.* 2015).

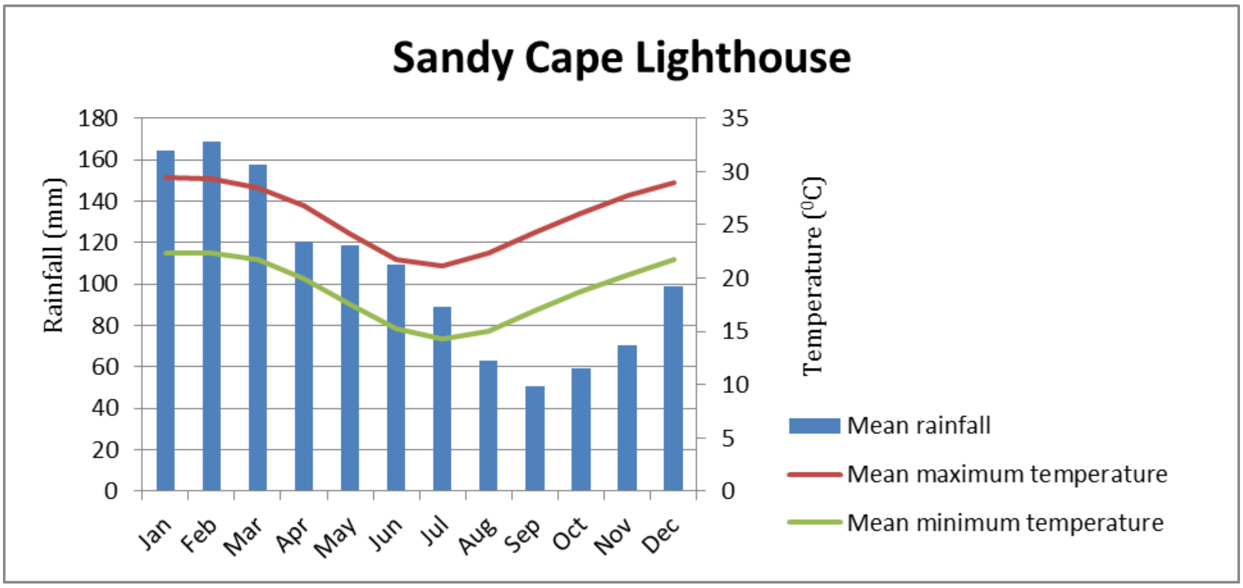


Figure 4.2. Mean monthly climate data for Fraser Island taken from Sandy Cape Lighthouse giving mean monthly rainfall in mm and mean maximum and minimum temperature in degrees Celsius. Bureau of Meteorology, 2016.

The vegetation is composed of mangroves, salt marsh, *Melaleuca* swamps, coastal scrub, *Empodisma minus* dominated wetland, *Syncarpia* wet sclerophyll forests, notophyll vine forests, dry sclerophyll forests/woodlands that are dominated by *Eucalyptus* and *Corymbia* species, to open diverse ‘Wallum’ heathlands (Srivastava *et al.* 2013). Vegetation of Fraser Island is laterally zoned following the contours of successive parabolic dune systems that run parallel to the islands coastline (Donders *et al.* 2006), with younger parabolic dunes to the east. Nutrient availability increases with the age of the sand and amount of detritus initially then declines due to retrogressive succession with ‘Wallum’ heath occupying the oldest dunes. (Walker *et al.* 1981; Longmore and Heijnis 1999; Donders *et al.* 2006).

Figure 4.3. An example of one of the sand blows found of Fraser Island. An extensive active transgressive dune field on the east coast of the Island north of Eurong. The transgressive dune field lies in the direction of the prevailing summer easterly and winter south-westerly winds.

Moon Point is situated on the west coast of Fraser Island and consists of mixed open sclerophyll forest/woodland and a variety of coastal and freshwater wetlands. The sediment core collected for this study was taken from close to the southern edge of a patterned fen (Figure 4.4), which is dominated by a mix of *Melalaeuca*, *Leptospermum* and heath shrubs and observes the presence of a grass/*E. minus* understorey. The depth of the core was 150 cm and composed of sand from 150 to 140 cm, lake muds from 140 to 78 cm, decomposed peat from 78 to 30 cm and fresh peat from 30 to 0 cm. Moss *et al*. (2016) have investigated the patterned fen development and found that *E. minus* (commonly known as the spreading rope rush; Restionaceae family) is the key peat forming species responsible for the formation of this wetland. Other common species include *Leptospermum liversidgei, Gahnia sieberiana, Hibbertia salicilfolia, and Sprengelia sprengelioides*. The patterned fen (and in fact most wetlands on Fraser Island) is highly acidic with a pH of between 4.3 and 4.8 which overlays the organic peat that vary in depths from 1 to 3 metres (Moss *et al.* 2016). Patterned fens (Figure 4.5), as well as the non-patterned *E.minus* wetlands, are subjected to fires that seem to play a role in reducing shrub cover. The vertical ascending *E. minus* roots are protected from fire and allow regrowth to occur and also appears to prevent the ignition of the underlying peat (Moss 2014; Moss *et al.* 2016). Fire is managed according to fire protection policy, biodiversity conservation and through fuel reduction activities (Stewart and Moss 2015; Moss *et al.* 2016).

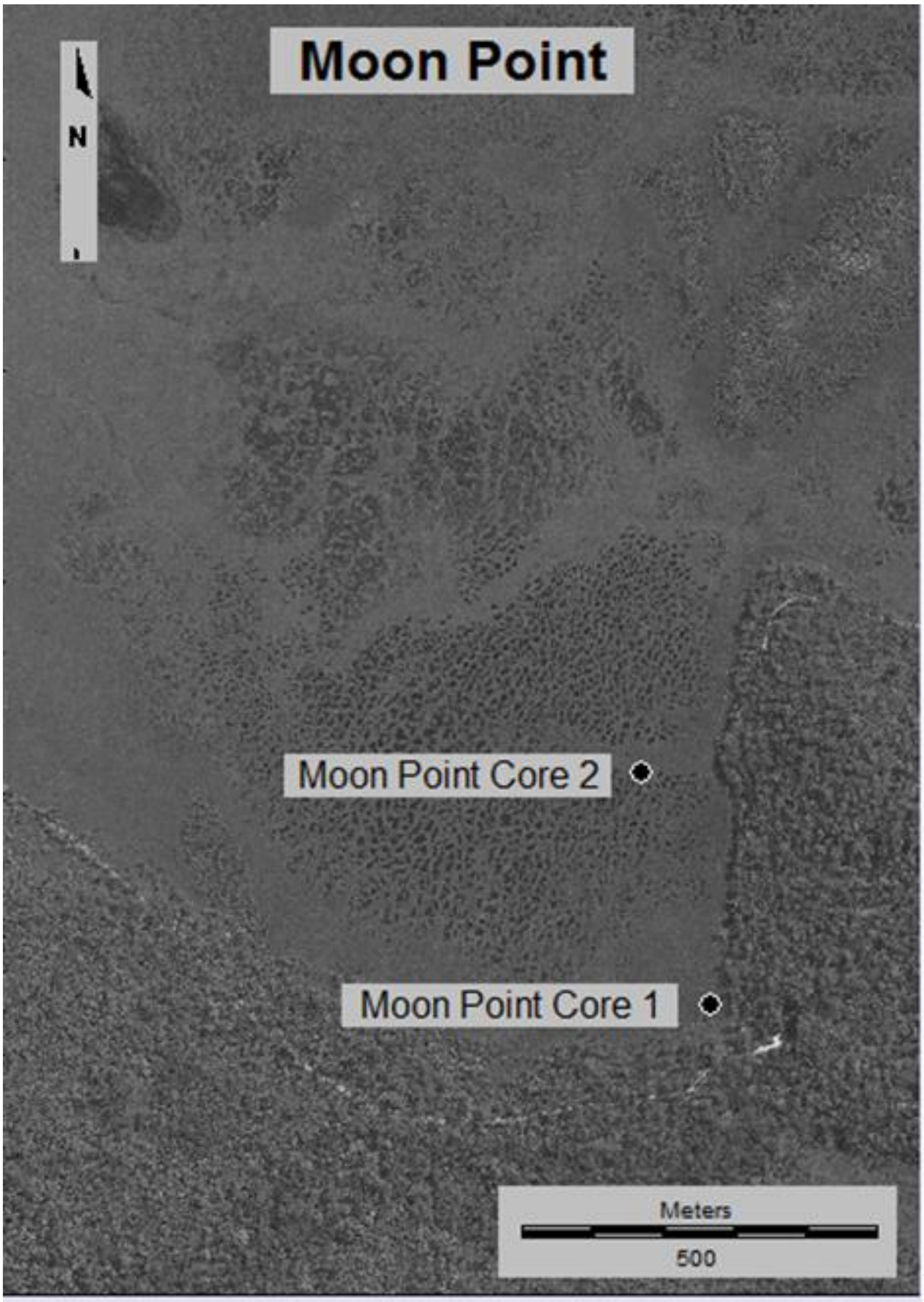


Figure 4.4. Location of the Moon Point cores that were collected in November 2013. Moon Point South is core 1 (lat 25^{o} 13' 11.89"S; long 153^{o} 03' 48" E) and Moon Point centre is core 2 (lat 25^{o} 13' 05" S; long 153^{o} 03' 42" E).

4.3 Methodology

4.3.1 Sampling

A 150 cm long sediment core was collected from close to the southern edge of the Moon Point patterned fen complex in November 2013, using a D-section (Russian) peat corer (Figure 4.5). The sample was transported to the Physical Geography Laboratory, School of Earth and Environmental Sciences at The University of Queensland. Macro charcoal analysis followed the Stevenson and Haberle (2005) methodology with 1 cm^3 samples collected at 1 cm intervals from the core. All samples were placed in beakers with 10% solution of sodium hexametaphosphate to disaggregate the sediment. The samples were then left overnight to settle and were then sieved through a 125 µm sieve. The samples were then transferred to 50 cm plastic test tubes and they underwent washing with distilled water. A 10% hydrogen peroxide solution was added to each sample to bleach all the organic material in the samples and left for 24 hours. The samples were then washed with distilled water again and stored in distilled water for macro charcoal counting. Samples were counted manually by placing each sample in Petri dishes over a one mm^2 graph paper which was placed under a Leica EZ4HD stereo microscope with 4.4:1 zoom. Analysis of macro charcoal involves the counting of particles of charcoal that were greater than 125 µm as large particles of charcoal are not usually transported far from fires (Whitlock and Larsen 2001; Sadori and Giardini 2007; Higuera *et al* 2009; Joannin *et al.* 2013; Sass and Kloss 2014; Inoue *et al*. 2015; Moss *et al.* 2015: Tan *et al.* 2015) providing evidence of the local fire history.

Figure 4.5. D-section (Russian) peat corer with core taken at Moon Point patterned fen in November 2013.

Pollen and micro charcoal (particles >10 to <125 microns) analysis was undertaken on the core at intervals of five cm and following the method developed by van der Kaars (1991) and described in detail by Moss *et al.* (2013). This analysis involved disaggregating one cm^3 subsamples in 10% sodium hexametaphosphate $(NaPO_3)_6$ and then using a heavy liquid solution (sodium polytungstate with a specific gravity of 1.9) to separate the lighter organic fraction from the heavier mineral component. The organic fraction, containing fossil pollen and the micro charcoal (<125 μm) fraction, then underwent acetolysis (9 ml acetic anhydride to 1 ml concentrated sulphuric acid and heated to 90°C for 6 minutes) to remove excess humic material and stain pollen grains for easier identification. The samples were then mounted on glass slides in glycerol and counted for pollen and micro charcoal using a Leica DM2500 compound microscope at 400 X magnification. Pollen and micro charcoal concentrations were determined using a *Lycopodium clavatum* tablet (exotic spike) with a known concentration (~20,848 spores). The pollen sum was based on at least 300 pollen or spores or two completely counted slides. TGView (Grimm 2004) was used to construct the pollen diagram and a stratigraphically constrained classification using CONISS (a feature of TGView, Grimm 1987, 2004) was used to determine zones based on the raw pollen counts.

4.3.2 Sediment Description and Age-Depth Models

The core sediments were described using a modified Troel-Smith system (Kershaw 1997) and the core also underwent moisture and organic content analysis. Moisture content was determined by weighing the samples before and after being placed in a drying oven (60°C for 24 hours) and LOI (Loss on Ignition) content was estimated from the mass lost after heating the samples at 490°C for 12 hours using a high temperature oven. Both the moisture and organic content results are shown with the pollen and micro charcoal data. Three samples (at 40, 65 and 145 cm) were taken from the Moon Point South core for AMS ^{14}C dating, which was undertaken at the University of Waikato and Beta Analytic (Table 4.1). Radiocarbon dates were entered into Bacon version 2.2 age-depth modelling software (Table 4.2), that uses Bayesian statistics to reconstruct Bayesian accumulation histories for deposits (Blaauw and Christen 2013) and the SHCal13 (Southern Hemisphere) Calibration curve was used for the conversion to calendar years Before Present (BP). The age model was also compared with a nearby sediment record (Moon Point Centre) that has an age model based on six dates and described in detail by Moss et al. (2016), with the sedimentological data from both sites used to confirm the age model for Moon Point South record.

Table 4.1. Radiocarbon dates for Moon Point South

Site/Depth	Dated material	^{14}C age (BP)	Cal years BP
Moon Point South - 40cm	Organic sediment	150 ± 15	Cal BP 24 to 258
Moon Point South - 65cm	Organic sediment	4570 ± 30	Cal BP 5310 to 5210
Moon Point South - 145cm	Organic sediment	19640 ± 60	Cal BP 23745 to 23485

Table 4.2. Core age and depth input for Bacon Age-depth model for Moon Point South.

ID	age	error	Depth cm	cc
Surface	150	15	40	3
2	4570	30	65	3
3	19640	60	145	3

4.3.3 Macro Charcoal Analysis

The age-depth model data were inputted into a stand-alone version of CharAnalysis (version 1.1) with the macro charcoal counts of each of the 150 samples. CharAnalysis is a set of diagnostic and analytical tools designed to analyse charcoal records in particular for the reconstruction of local fire history (Higuera 2009). Peaks in charcoal accumulation rate (CHAR being number of pieces of charcoal per square centimetre per year) have shown both empirically (Lynch *et al.* 2004; Ali *et al.* 2009 and Senici *et al.* 2015) and mechanistically to be associated with local fire events in lake sediment records (Higuera *et al.* 2007; Higuera *et al.* 2008).

Time-series analysis of the charcoal data was undertaken using CharAnalysis (Higuera *et al.* 2009) to identify charcoal peaks as positive residuals exceeding the 99th percentile threshold of the locally fitted Gaussian mixture CHAR background model, which is a robust statistical proxy for local fire events smoothed to 500 years (Fletcher *et al.* 2014).

Peaks in the charcoal accumulation rate (pieces cm^{-2} yr^{-1}) identify local fires events within a 1–3 km radius and total charcoal accumulation reflecting area burned at larger spatial scales (Higuera *et al*. 2011). Charcoal data were first interpolated to 100 year intervals, based on the average temporal resolution of the records. The low-frequency CHAR, $C_{background}$ represents changes to secondary transport, sedimentation, mixing and charcoal production were removed to estimate the timing of fire events in the charcoal records (Higuera *et al.* 2008), leaving residual series C_{peak} ($C_{peak} = C_{interpolated} - C_{background}$). C_{peak} values have two sub-populations, C_{noise} and C_{fire}, where C_{noise} is a normally-distributed population centred near 0 and where C_{fire} samples are high CHARs exceeding variations in C_{noise} (Higuera *et al.* 2008; MacKenzie 2016). C_{noise} is composed of variability from sediment mixing, sampling, natural and analytical noise and was calculated using a Gaussian mixture model, with 95th, 99th and 99.9th percentiles as thresholds to identify fire events. $C_{background}$ was modelled using a robust locally weighted regression with a 250 year window to maximise the signal-to-noise index and C_{noise} goodness-of-fit (MacKenzie 2016). Peaks exceeding the locally defined threshold were assessed and compared with the original charcoal counts that contributed to each peak where any CHAR peak had a >5% chance of being from the same Poisson-distribution population as the minimum charcoal count within the prior 75 years were excluded (Higuera *et al.* 2008; MacKenzie 2016).

4.4 Results

4.4.1 Age-Depth Bayesian Model

Bacon produced an age-depth model with an output graph depicting the MCMC iterations with 5 panels plotted with the Bacon age-depth model (Figure 4.6) within the core folder. A text file MP_110_ages.txt was produced that contained the 95% age ranges, median and weighted mean for each centimetre depth. In the same folder were produced a settings text file giving the applied settings, MP_110.out with the parameter values that made up the age-model and a MP_110.bacon file that contained the chosen commands to produce the age-model.

Bacon simulation run produced a Bayesian model with median years BP 150 at 40 cm depth, based on the suggested date for European arrival from exotic pollen and a corresponding age from a core taken from the centre of the wetland (Moss *et al*., 2016) to years BP (Before Present) 25,182 at 150 cm depth, suggested from the basal radiocarbon age (Table 4.1).

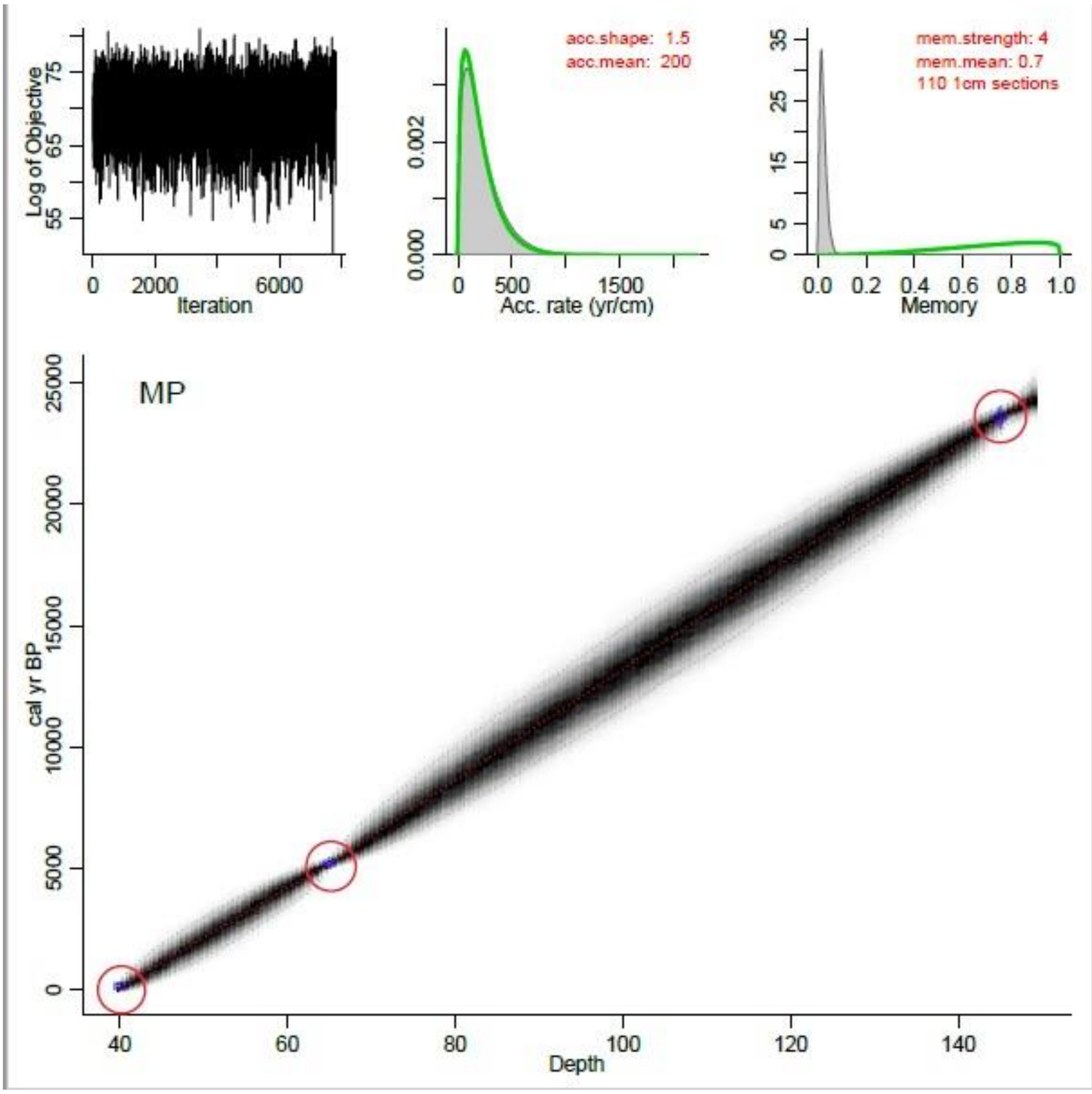


Figure 4.6. Bacon age-depth model output graph with upper left panel depicting the Markov Chain Monte Carlo (MCMC) iterations for the age-depth model with good runs and a stationary distribution with little structure between neighbouring iterations. Accumulation rate (second panel from left) and Memory (third panel from left) show the prior (green curve) and posterior (grey histograms) distributions for the accumulation rate. The main panel gives the calibrated ^{14}C dates (blue dots encircled in red) and the age-depth model (darker grey), with the grey stippled lines showing 95% confidence intervals. Red line shows single best model based on weighted main age for each depth (Blaauw and Christen, 2013).

4.4.2 Macro charcoal Analysis and Fire History Reconstruction

The Bacon age-depth model was placed in CharAnalysis (charDATA) for analysis of local fire histories of the Moon Point South core sample to analyse the macro charcoal for fire return intervals, fire frequency and charcoal peak magnitude (Higuera 2009). The timing of local fires was inferred by decomposing charcoal records to identify peak charcoal based on a standard set of thresholds applied to all records. Default parameters for CharAnalysis were tested with the initialisation and execution of the model and were found to be robust for the analysis of the Moon Point macro charcoal. In CharAnalysis the charcoal data were decomposed to distinguish between background charcoal from distinct charcoal peaks based on a set of threshold criteria (Higuera *et al.* 2007; Higuera *et al.* 2008; Maezumi *et al.* 2015). All peaks identified were screened to eliminate peaks when variations in CHAR based on small differences in charcoal counts (Higuera *et al.* 2009). The median signal-to-noise index (SNI) for all records where peak analysis was interpreted was 1.10, with the SNI for all records mostly >0.5. Each histogram (Figure 4.7) represents a 500 year non-overlapping section of the record. All samples above the threshold value t_i (red vertical line) represent charcoal from local fires (C_{fire}). The threshold cuts out 99% of the samples assumed natural and analytical noise (C_{noise}) distribution (blue line) and is repeated for each and every sample of the record, providing a unique threshold value giving the variability in C_{noise} for each sample. The signal-to-noise index (SNI), t_i and C_{noise} vary throughout the record. KS *p* is the *p* value resulting from a two-sample Kolmogorov-Smirnoff goodness-of fit test between the empirical C_{noise} values and the modelled C_{noise} distributions proving an index of how well the C_{noise} model fits the empirical data. Figure 3.8 shows the macroscopic charcoal results from Moon Point analysed with CharAnalysis (Higuera *et al*., 2009). i) Shows the raw charcoal accumulation rate over interpolated 100 year intervals, and the background signal (Lowess smoothing robust to outliers 500 year trends). ii) Shows peak CHAR Cpeak with red lines (both negative and positive) identifying noise related variables and peaks are identified with each threshold criteria or plus sign (+). The peaks that failed to pass the Poisson minimum-count are identified by gray dots. iii) Shows $C_{interpreted}$ and $C_{background}$ thresholds with peaksFinal as the plus sign (+) with peaks 1 and peaks 2 plotted as gray dots. iv) show a Boxplot of all SNI values.

Moon Point South results show that charcoal accumulation rate peaked between 24,500 and 22,500 cal. yrs BP dropping off to a level rate of charcoal production until a peak in accumulation occurred at 19,000 cal. yrs BP. This was followed by a drop between 18,500 cal. yrs BP to 11,500 cal. yrs BP in and levelling of charcoal occurred, which was followed by a number of peaks in accumulation rates in11,500 cal. yrs BP, 9400 cal. yrs BP, 7500 cal. yrs BP, 5500 to 4500 cal. yrs BP, 3500 cal. yrs BP, and 2500 to 1500 cal. yrs BP. From 1500 cal. yrs BP to1800 AD charcoal accumulation rates remained low, however from 1820 to 1900 AD there is an increase in accumulation rate that may correspond to the European settlement period.

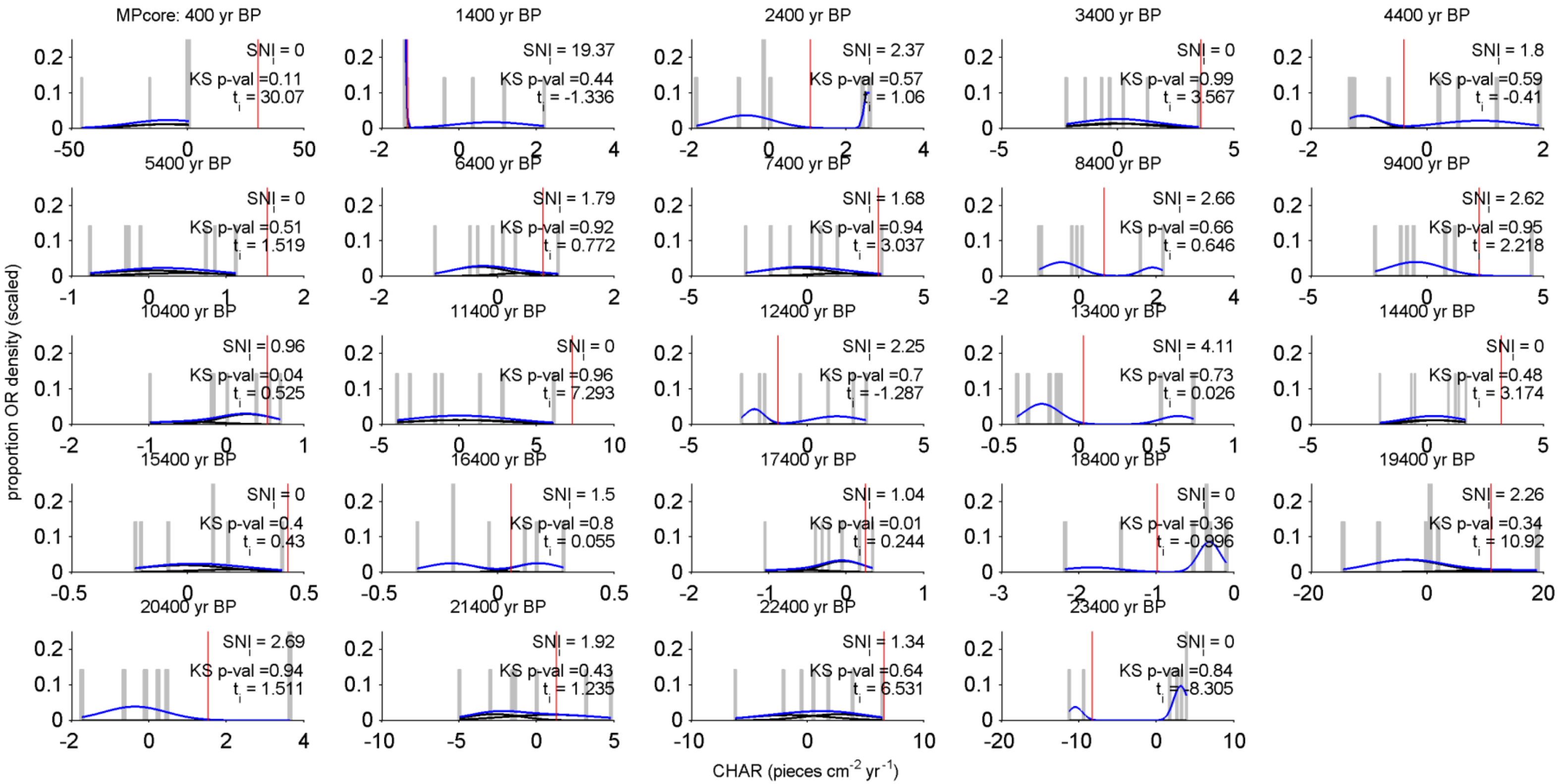


Figure 4.7: CharAnalysis results from Moon Point South core. Local distributions (Histograms) of the peak charcoal series C_{peak} from Moon Point South core showing multiple non-overlapping 500 year time periods spanning the charcoal record, including two modelled Gaussian distributions. All samples above the threshold value t_i (vertical red line) represents charcoal from local fire C_{fire}. The sample cuts off 99% of the samples assumed to represent natural and analytical noise, C_{noise} distribution (blue line with lower mean). KS *P* gives the *P* value of the two-sample Kolmogorov-Smirnoff goodness-of-fit test, providing the index of how well the C_{noise} model fits the empirical data. The distribution of C_{noise} varies throughout the record as does t_i and the SNI.

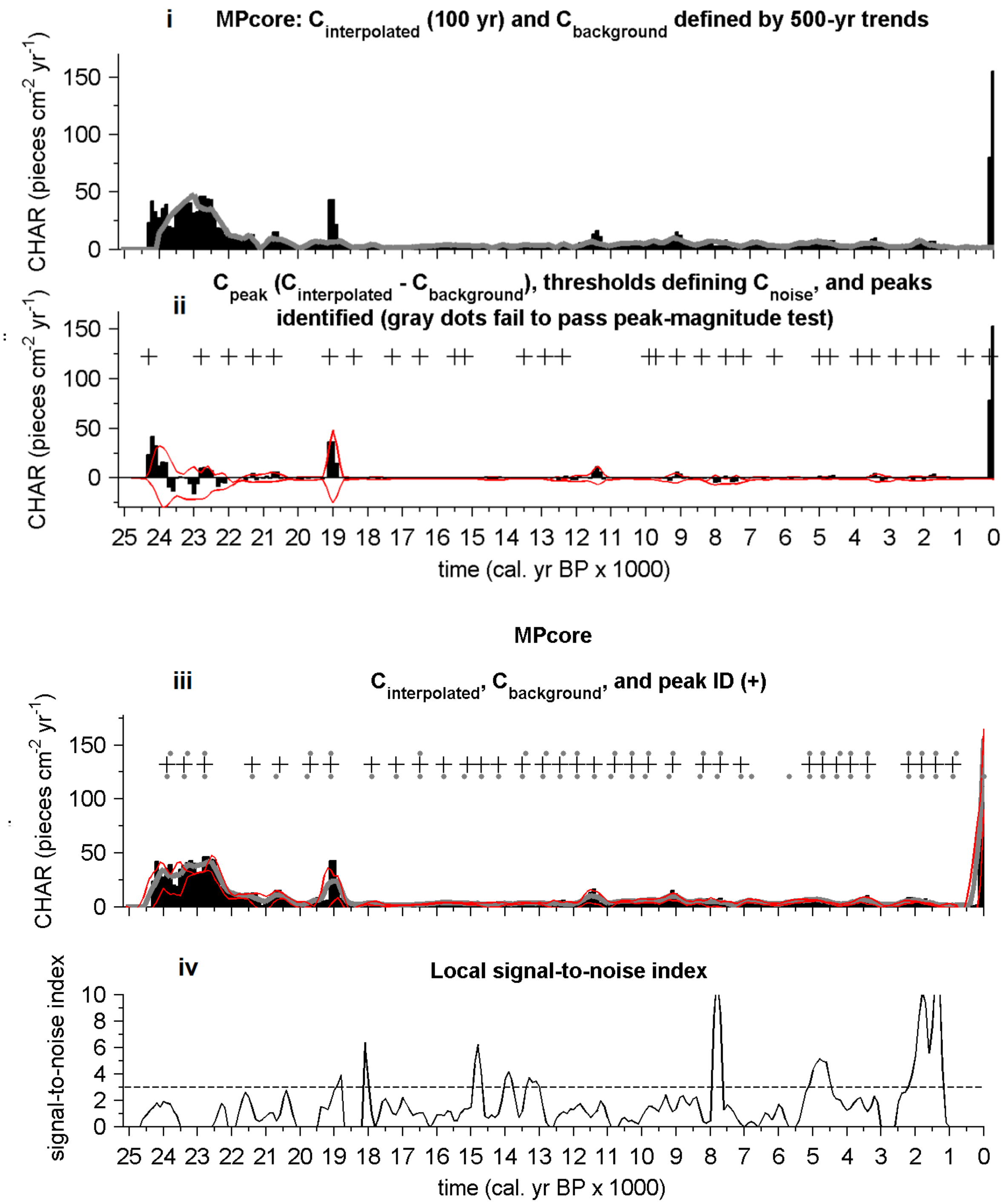


Figure 4.8: i) and ii) Interpolated charcoal and low frequency trends with de-trended series (C_{peak}). Iii) and iv) show sensitivity to alternative thresholds and quality of the records for Moon Point South core.

4.4.3 Pollen and Micro Charcoal Analysis

A total of 62 pollen taxa have been identified through analysis undertaken by Moss of the Moon Point South pollen record, which have been grouped into rainforest, sclerophyll arboreal taxa, sclerophyll herbs, aquatic taxa, pteridophytes and mangroves based on the habit of the parent plant taxa (P. Moss 2016, pers comm., 2nd May). Figure 4.9 shows a summary of the key pollen taxa, along with the pollen and charcoal concentration data, as well as the age model for the record. Also, included is a dendrogram based on the cluster analysis undertaken on the raw pollen count data and this has identified six zones, which are discussed in detail below [including depth and age in calendar years Before Present (cal. yrs BP)].

MPS 1 (150 to 145 cm; ~24,260 to ~23,690 cal. yrs BP)

This zone observes the highest abundances of rainforest taxa (~60%), predominantly Cunoniaceae, *Elaeocarpus*, Myrsinaceae and *Argyrodendron*, and with smaller values *Olea paniculata*, palms and *Macaranga/Mallotus*. Sclerophyll arboreal taxa are the next most important group, which is dominated by *Callitris* (~20%) and with only low values of the other groups (i.e. herbs, aquatics, pteridophytes and mangroves). Pollen concentrations range from 100,000 to 300,000 grains per cm^3, while the micro charcoal values are below 100,000 particles per cm^3. The moisture content ranges from 50 to 60% and organic content is around 30%, reflecting the sandy basal units of this zone.

MPS 2 (140 to 105 cm; ~22,560 to ~14,450 cal. yrs BP)

There is a marked decline in rainforest taxa in this zone, with their virtual elimination from the record at the top of this zone, Myrsinaceae and *Elaeocarpus* are the first taxa to decline and subsequently followed by the remaining taxa, *Callitris* also declines. Sclerophyll arboreal taxa, mainly *Melaleuca* and Casuarinaceae from 120 cm (~17,620 cal. yrs BP), markedly increase, while there is also a clear increase in aquatic taxa abundances (primarily Restionaceae, *Hibbertia* and *Hydrocotlye*) and a peak in *Myriophyllum* at 105 cm. Grass also increases in this zone and there is an interesting peak in mangroves in this zone as well. Pteridophytes maintain very low values in this zone and for the rest of the record. Pollen concentrations are generally below 200,000 grains per cm^3, while micro charcoal concentrations are below 100,000 grains per cm^3 throughout this zone. The sediment changes to organic silt, with moisture content generally above

60% and increasing towards the top of the zone and with the organic content still around 30%, although there are two sharp peaks at 130 and 110 cm.

MPS 3 (100 to 65 cm; ~13,300 to ~5770 cal., yrs BP)

This zone observes a dramatic increase in Restionaceae values (to ~60%) from 100 to 90 cm (~13,300 to 11,000 cal. yrs BP), this aquatic taxon then declines to around 20% by the end of the zone, with a corresponding increase in *Myriophyllum*. There is a clear increase in both moisture (above 75%) and organic content (above 70% but does decline from 80 cm), which corresponds to the increased Restionaceae abundances and the transition to a peat sediment. The other key changes in this zone are a sharp decline in *Melaleuca* and a slight increase in the heath taxa (Ericaceae and *Monotoca*), while the Casuarinaceae maintain consistent abundances (~20%) throughout this zone and in fact for the rest of the record. In addition, there is a marked increase in pollen (generally above 400,000 grains per cm^3) and micro charcoal (generally above 100,000 particles per cm^3) in this zone.

MPS 4 (60 to 55 cm; ~4280 to ~3240 cal. yrs BP)

A marked peak in mangroves (primarily the Rhizophoraceae) and *Melaleuca*, with a corresponding sharp decline in *Myriophyllum* and a smaller decrease in Restionaceae values seen in this zone. This corresponds to a sharp decline in moisture and organic content, although there is no obvious change in the peat sediment in this zone. There is a minor peak in Cyperaceae, while eucalypt abundances increase at 55 cm to around 5% and maintain these values for the remainder of the record. Pollen and micro charcoal concentrations are consistent with the previous zone.

MP 5 (50 to 25 cm; ~2190 cal yrs BP. to 1923 AD)

A recovery (to ~50%) in Restionaceae, as well as moisture and organic content is observed in this zone, while mangroves virtually disappear and *Melaleuca* sharply declines, there is also a small decline in grass values. There is a small increase in *Callitris* values and there is a minor peak in *Banksia* at 35 cm (1882 AD). There is a dramatic peak in pollen (above 600,000 grains per cm^3) and micro charcoal (above 7 million particles per cm^3) concentrations observed at 25 cm. There is a change in sedimentology at 30 cm (~1900 AD) from decomposed peat to fresh peat.

MP 6 (20 to 0 cm; 1938 to 2013 AD)

Restionaceae abundances markedly decline in this zone to around 30%, while there is a corresponding increase in Myrtaceous shrubs, *Callitris*, heaths, Cyperaceae, *Hydrocotyle* and mangroves. There some variability in in pollen and micro charcoal values in this zone, with a clear peak in micro charcoal abundances seen at the top of the record and pollen concentration abundances are lower. Moisture and organic content are similar to the previous zone.

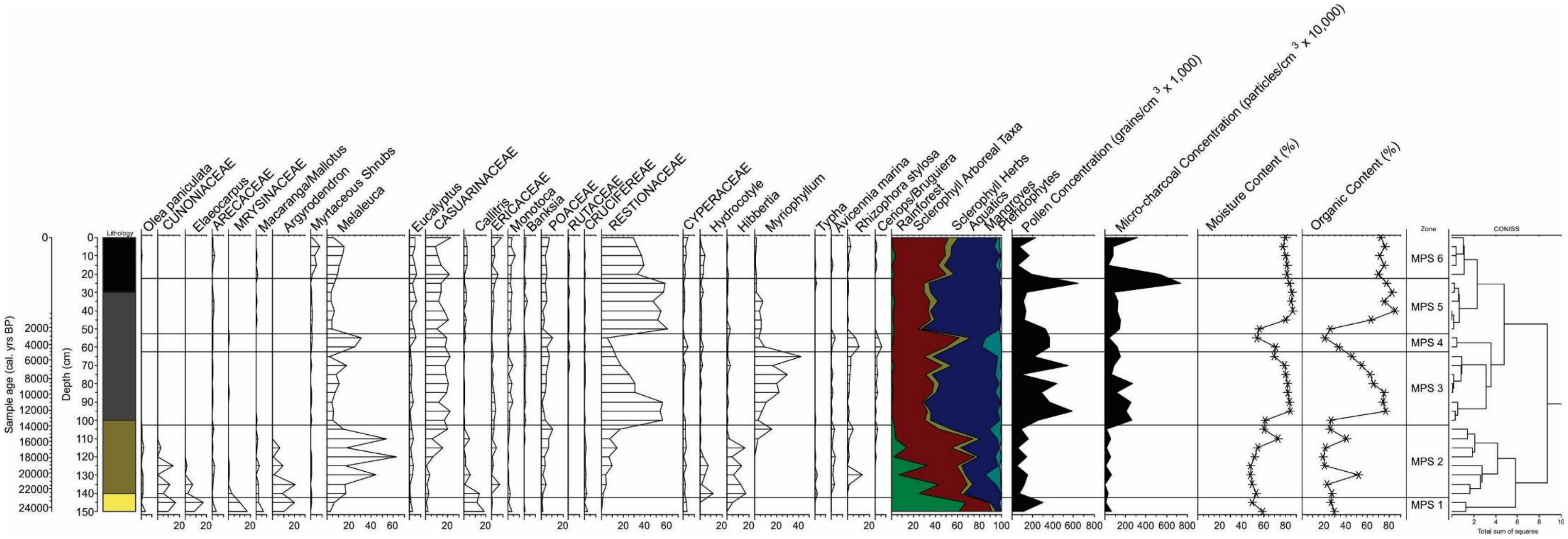


Figure 4.9: Full pollen diagram of Moon Point South plotted against depth and age showing vegetation taxa, pollen concentrations, micro charcoal concentrations, moisture and organic content.

4.4.4. Comparison of Micro and Macro charcoal

Micro and macro charcoal record variability of the extra-local and local fire events for Moon Point South since the Last Glacial Maximum 24,500 cal yrs BP. Charcoal is present throughout all records with relative low levels of micro charcoal initially from 24,000 to 13,300 cal yrs BP. However during the same period high levels of macro charcoal are recorded, which may be linked to a drying climate (Moss *et al.* 2013), and anthropogenic fire. From 13,300 to 2190 cal yrs BP micro and macro charcoal levels remain fairly constant. Micro charcoal peaks suddenly at 25 cm, with macro charcoal peaking at 26 cm, both peaks coincide with the possible arrival and settlement of Europeans on the Island in 1840's (Lennon 2012), when charcoal deposits increase due to possible land clearing and modification of the landscape (Stewart and Moss 2015). However the quantity of charcoal has no clear correlation to either size or frequency of fire but may reflect fuel, for example dry timber may proceed to ash while moist grass may produce copious char particles.

4.5 Discussion

A number of factors influence the type and amount of charcoal being produced by a fire such as the type of materials being burnt (grasses and trees) including intensity and duration of fire play an important role in charcoal production (Patterson *et al*. 1987). Charcoal records are composed of primary (local fire event) and secondary sources (introduced later through run-off or redeposition), therefore estimating fire size, severity, or intensity is possible only in general terms (Whitlock and Larsen 2002).

4.5.1 Local Fire and Vegetation History of Moon Point since the Last Glacial Maximum

The detailed analysis of macro charcoal had found that fire events have occurred throughout the record through the identification of charcoal peaks above the 99% threshold level indicating that fires had occurred within a 3 km radius of Moon Point South site (Higuera *et al.* 2011). The charcoal peaks represent fire events whereas variation in the frequency of peaks represents changes in the occurrence of fire events through time (Whitlock *et al*. 2004).Macro charcoal records show during 24,500 to 22,000 cal. yrs BP fire events occurred on a local scale with relative higher frequency at Moon Point South and within a 3 km radius, with the levels of CHAR representing area burnt (Figure 4.8.iii). Micro charcoal levels are below 100,000 grains per cm^3 from 24,260 cal. yrs BP to approximately 14,450 cal. yrs BP indicating fire activity was present but at low levels at a regional scale, with macro charcoal suggesting higher fire activity (higher frequency) and more area burning at a local level for the period 24,500 to 22,000 cal. yrs BP, followed by variations in macro charcoal accumulation rates until 19,000 cal. yrs BP, when there was another peak in local burning (Figure 4.9). These peaks, along with the sharp decline in rainforest and increase in sclerophyll taxa suggests a response to the dryer climates of the Last Glacial Maximum period (Petherick *et al.* 2013; Moss *et al*. 2013), which appears to be impacting the local environments around the core site. A similar trend (i.e. decline in rainforest and increase in sclerophyll taxa) is observed from a core taken from the central portion of the Moon Point mire at a similar time period (Moss *et al*. 2016). However, the macro- and micro charcoal trends are reversed, with evidence of more regional burning (micro charcoal) and less sustained local burning, although there are some large peaks in macro charcoal. The differences in burning trends may reflect the different settings of the core sites, with this studies site located on the edge of a lake and receiving direct input of large charcoal fragments but receiving less of the regional signal. In contrast, the core site discussed in Moss *et al*. (2016) was situated in the centre of the lake, receiving a greater input from regional fires. Moss *et al*. (2013) have discussed the importance of geographic setting for micro charcoal representation on North Stradbroke Island wetlands and a similar trend seems to be apparent for Fraser Island, which warrants further investigation.

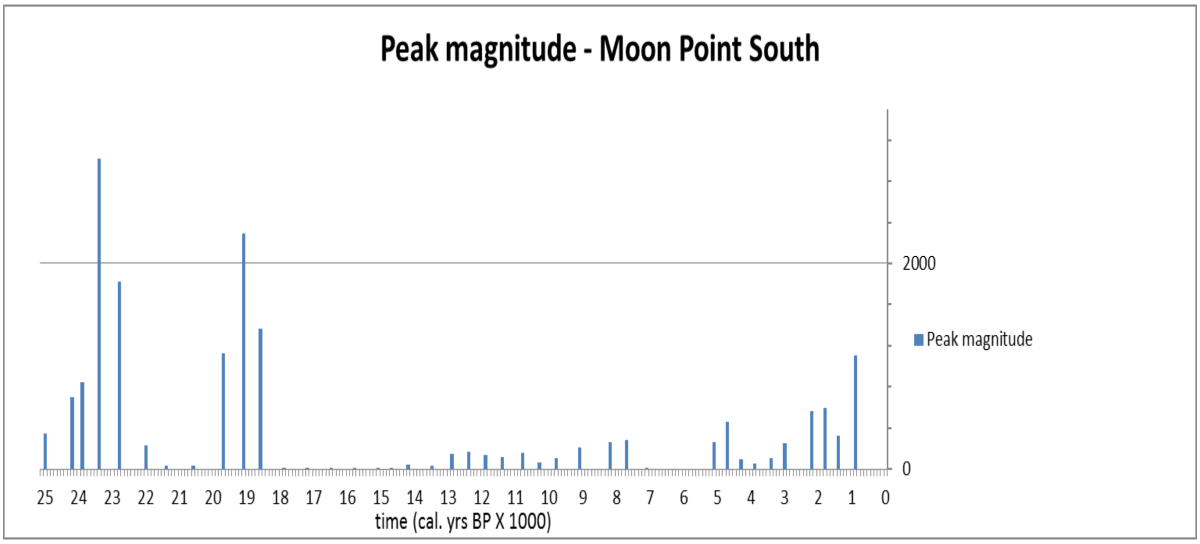


Figure 4.10: Peak magnitude charcoal showing fire size, severity and proximity for Moon Point South.

At approximately 14,600 cal. yrs BP an increase in aquatic abundances (primarily Restionaceae, *Hibbertia* and *Hydrocotyle*) occurs and a peak in *Myriophyllum,* which may reflect the development of a shallower lacustrine system. Grass also increases in this zone and there is an interesting peak in mangroves, which may be linked to changes in sea level. The post-LGM sea-level rises occurred from 16,000 to 12,500 years ago and again from 11,500 to 9000 years ago, the interval between these two periods corresponding to the Younger Dryas, (Lambeck *et al.* 2001).

There is an increase in CHAR around 13,000 cal yrs BP, which continues to rise until approximately 7,600 cal. yrs BP suggests higher fire activity and probably increases in area burnt, followed by a rapid drop in CHAR from approximately 7,400 cal. yrs BP to 5,400 cal. yrs BP. At the same time (13,000 to 7,600 cal. yrs BP) there is a dramatic increase in Restionaceae values (~60%), which declines to around 20% by 5770 cal. yrs BP. There is a corresponding increase in *Myriophyllum* at this time, which may indicate the presence of a shallow lake (Barrett *et al.* 2016). There is a sharp decline in *Melaleuca* corresponding with an increase in CHAR and fire frequency around 5,070 cal. yrs BP. With this there is a slight increase in heath taxa, Ericaceae and *Monotoca*. The Casuarinaceae maintain consistent abundances throughout this period. From approximately 7,400 to 5,100 cal. yrs BP levels of CHAR remain low

A marked increase in CHAR level occurs around 5,000 cal. yrs BP that increases to 3,600 cal. yrs BP before dropping off steeply that corresponds with the marked peak in mangroves (primarily the Rhizophoraceae) and *Melaleuca*, with a corresponding sharp decline in *Myriophyllum* and a smaller decrease in Restionaceae values. This may be due to earlier sea-level rise, a drying climate and an increase in anthropogenic fires. Permanent occupation by Aboriginals occurred approximately 5,000 years ago on Fraser Island (Lennon, 2012).

CHAR levels increase again from 3,300 to 3,000 cal. yrs BP indicating an increase in fire activity and possibly area burnt and degree of vegetation mortality. From 2,190 cal. yrs BP to 800 AD CHAR levels substantially increase reflecting peak magnitude in fire size, severity and proximity (Figure 4.10), which is linked to with an increase in Restionaceae, with the virtual disappearance in mangroves and *Melaleuca* sharply declining over this period. There is also a substantial increase (peak) in micro charcoal concentrations observed at approximately 1850 to 1930 AD. After this period there is a marked drop in CHAR to 2013 AD which indicates a sharp change in fire activity and reduced fire frequency, which may correspond to a fire exclusion policy for the island (Kershaw *et al.* 2002; Stewart and Moss 2015).

4.5.2 Regional Fire History and Vegetation Change

The micro charcoal analysis (Figure 4.9) provides an extra-local regional picture of fire since the LGM (Mackenzie 2016) and in combination with the pollen record from the southern edge of the Moon Point mire, as well as the previously published 40,000 year record from the central part of this wetland (Moss *et al.* 2016) provides insight into Late Quaternary environmental change for the subtropics of coastal eastern Australia.

Quaternary sea-level fluctuations have had a substantial impact on the distribution of mangroves (Rhizophoraceae) on both the local and regional scale with mangrove expansion and contraction occurring during the Holocene (Hashimoto *et al.* 2006). It has been suggested by Clarke and Guppy (1988) that the high number of mangrove in the LGM could be the reworking of mangrove pollen from Marine Isotope Substage (MIS) 5e high-stand (last interglacial) sediments or that mangroves were growing up river during low sea-level periods reflecting estuarine mangroves (Clarke and Guppy 1988). It is interesting to note high mangrove pollen values at Moon Point south (Figure 4.9) during the LGM and this is rather an unexpected result due to the dramatically lower sea-levels of this period (-120 m) and the distance (>40 km) from the coast.

Longmore (1997) undertook pollen and charcoal analysis from Lake Coomboo, a perched lake on Fraser Island that provided regional Quaternary palynological records for the area showing that *Araucaria* species pollen levels increased during the Last Interglacial and before the LGM before falling rapidly. Longmore (1997) found that open forest shifted to myrtaceous shrubs on the west of the Fraser Island before the LGM and that open forest returned post LGM and that *Araucaria* never recovered. According to Donders *et al.* (2006) pollen analysis of a sediment record taken from Lake Allom there is evidence of a change from rainforest and open woodland, with dry conditions in the early Holocene to higher lake levels with increased forests between mid to late Holocene. Charcoal records show increases in fire events during the periods of drier vegetation and low lake levels. During the same time, North Stradbroke Island experienced similar conditions with a wetter climate in the early to mid-Holocene, with lower charcoal levels that suggest less fire events compared to the late Holocene when the climate became drier, which is marked by an increase in charcoal levels and an increase in fire events (Moss *et al.* 2013). This is supported by the pollen and charcoal data of Moon Point south, showing similar trends with rainforest taxa and shift to sclerophyll vegetation with increased fire events (Figure 3.9).

4.5.3 Fire and *Empodisma minus* Wetlands

Many Restionaceae do not reach full reproductive ability for the first 4 to 8 years after a fire therefore having a low reproductive ability and as obligate seeders can become locally extinct if periods between fires are too short (Meney *et al.* 1994). *E. minus,* a key wetland species (Wagstaff and Clarkson 2009), are resprouters and fire plays an important role in the development of restiad peat fens where the underground parts of *Empodisma* are protected and survive fire (Clarkson 1997). *Empodisma minus* respouts within weeks of a fire and recovers rapidly and can reach pre-fire levels within 2 years of burning and will continue to grow considerably for up to 17 years post fire (Wahren and Walsh 2000; Wagstaff and Clarkson 2009). Changing fire regimes will impact on the ability for particular heathland species and especially non-dicotyledons such as *E. minus* to remain dominate, given that fire impedes successional processes that convert wetlands into forest (Driessen and Kirkpatrick 2017), therefore allowing woody species to take over dominance (Watson 2001). Fire is important for the maintenance of *E. minus*, as a decrease in frequency will see above ground growth increase resulting in more intense fires that in turn will favour surrounding forests and potentially allow them to invade the wetland. Therefore, low intensity but frequent fire is needed to help maintain the extent of *E.minus* dominated wetland preventing encroachment and vegetation thickening (see Chapter 5). This is clearly seen in the pollen record (Figure 4.9) in MP5 and MP6 there is a marked increase in micro charcoal and a corresponding decrease in *E. minus* pollen. Further changes in sea-level at Moon Point south have also influenced *E. minus,* first around 10 to 12 ka years ago and then severely impacting it around 5 ka during the highstand in sea-level suggesting a complex relationship.

The pollen data suggests an increase in woody taxa and an increase in macro charcoal levels suggesting thickening has occurred since fire suppression and arrival of Europeans on the island. Presently fire management in the Great Sandy National Park uses prescribed burning to promote biodiversity and reduce wildfire hazards (Moss *et al.* 2016). *E.minus* is a non woody species that is associated with peaks in micro charcoal suggesting broad scale burning in the region (Moss *et al.* 2016), where macro charcoal peaks are associated with the burning of woody material and local fire events. Moon Point records show that fire occurs within the *E minus* wetlands every seven to 15 years on average since 1977 (Moss 2014). These fires would have been patchy as a result of the wet and often damp vegetation of *E.minus* wetlands as identified in a recent prescribed burn of Moon Point in June 2016 (L. Behrendorff 2016, pers comm., 24 June). According to aerial photographs and land change analysis from 1958 to 2010 there appears to be forest vegetation thickening and encroachment onto the *E.minus* wetlands. Quantitative analysis of the aerial photographs of 1958 to 2010 show gains of forest over losses to Empodisma wetlands. Data show that forest has gained approximately 1.53% and the Empodisma wetlands showing a loss of 2.07%. These gains and losses are supported by the results of a land change model analysis of the aerial photographs (see Chapter 5).

4.6 Conclusion

This chapter discusses macro micro charcoal and pollen analysis of a sediment core taken from the edge of the Moon Point wetland. It explores the changing environment from the LGM, early to late Holocene and the changes in vegetation dynamics of Moon Point South with a special focus on paleofire regimes, climate, sea level and anthropogenic fire activities. Over the past 25,000 years Moon Point has seen sequential fires that have been driven by climate change and anthropogenic fire regimes.

Macro charcoal and pollen records show dramatic changes in vegetation composition and structure changing from rainforest taxa to sclerophyllous vegetation, driven by fire, climate and sea-level change and possible anthropogenic fires early in the records. During the last 5000 years the record show a sudden increase in fire events through the increase in charcoal peaks, which coincides with permanent settlement of the island by Aborigines around 5 ka and the intensification of ENSO, increased burning and associated vegetation alterations (Gontz *et al.* 2015). At 1840 AD the record shows a substantial peak in charcoal levels which drops off rapidly coinciding with the arrival of Europeans on the island and the establishment of forestry on the island. The increase in fire events during the early European settlement phase is thought to be due to burn-offs and clearing of unwanted vegetation, with the decrease in burning associated with the later development of a fire exclusion policy for forest product protection which has been followed through to 1991 until the island was handed over to become part of the Great Sandy Regional National Park. Since then prescribed fires have been implemented for biodiversity and hazard reduction.

Changing fire regimes have seen changes in vegetation composition and structure, leading to thickening of forest and encroachment of woody taxa on the Empodisma wetlands, which is discussed in more detail in Chapter 5.

Chapter 5. Land Change Analysis of Moon Point Vegetation, Fraser Island – Analysis of aerial photographs 1958 to 2010 and 2010 to 2016.

Abstract

This study investigates vegetation thickening and encroachment at Moon Point wetlands on Fraser Island using historical aerial photographs with present day data and a land change modeller to identify the extent of thickening and encroachment of three major vegetation types, forest, Banksia and wetlands. Through quantified assessment of the aerial photographs using fine-gauged graticules vegetation types were manually assessed and counted to identify changes in woody vegetation density. Further vegetation polygons were created of the three vegetation types for analysis through a Land change modeller for change analysis to identify percentage gains or losses of vegetation types, transition potential for change and change prediction to fifty years into the future. Both the quantified assessment and land change modeller identified vegetation thickening with percentage gains for forest and Banksia, with forests having the greatest percentage gains. Wetlands showed an overall percentage loss to forest and Banksia, which increases fifty years into the future.

5.1 Introduction

Fire is a driver of vegetation variability (van Wilgen 2009) and any changes to the fire regime impacts these dynamics that may result in significant ecosystem changes (Bond and Keeley 2005). In particular, it is a major ecological driver for the production of pyrogenic vegetation that is associated with the development of coppicing and serotiny mechanisms to survive the impacts of fire (Russell-Smith *et al*. 2001). However, regardless of the importance of fire in shaping the vegetation dynamics (i.e. structure and patterns), it is still poorly understood as to how frequency, season, intensity and sequence of fire interact in influencing vegetation structure (Smit *et al.* 2010). According to Higgins *et al.* (2007) tree density is not influenced by fire, only the size structure and biomass of trees are affected through fire frequency but the precise response of vegetation to a prevailing fire regime is not readily known.

Vegetation thickening is known to occur when fire has been withheld for years in sand environments (Vigilante and Bowman 2004). Edwards *et al.* (2003) found that with an increase in woody thickness there was a corresponding decrease in cover and species richness of herbs with areas experiencing lower frequency of fires. Bradstock *et al.* (1996) found that obligate seeder heathland species such as *Banksia* became locally extinct when subjected to infrequent fires due to senescence. Further, fire season and intensity are important as it has been shown that cooler earlier fire season fires favours and encourages woody vegetation and thickening (Smit *et al.* 2010). For instance, in tropical northern Queensland pastoralists have changed the fire regime by withholding fire for several years and burning only in the early dry season resulting in a loss of grasslands while increasing the woody vegetation (i.e. expansion in *Melaleuca viridiflora*) (Crowley *et al.* 2009). According to Russell-Smith and Edwards (2012) canopy thickening is negatively correlated with fire frequency.

Within heathlands in the Sydney sandstone region and alpine heaths it has been recorded that fire stimulates and increases vegetation density that levels off after approximately 10 to 15 years post fire (Keith *et al.* 2002; Walsh and McDougall 2004). For example *Empodisma minus* (Restionaceae)*,* a key wet heathland species on Fraser Island and a rhizomatous perennial (Wagstaff and Clarkson 2009), increases in growth significantly within the first 2 years after fire, with density levelling off approximately 6.5 years post fire (McFarlane 1988). Therefore changing fire regimes may impact on the ability for particular heathland species and especially non-dicotyledons such as *E. minus* to remain dominant, allowing for possible forest or woodland species to invade wet heathlands (Watson 2001). According to Moss (2014) *E. minus* is an ecosystem engineer responsible for creating fens and bogs where it forms dense mats of horizontal rhizomes holding water up to 15 times their dry weight, and is an important component of both patterned and non-patterned mires on giant sand masses of subtropical eastern Australia (Moss *et al.* 2016).

Many Restionaceae do not reach full reproductive ability until 4 to 8 years post fire therefor having a low reproductive ability and as obligate seeders can become locally extinct if periods between fires are too short (Meney *et al.* 1994). *E. minus* are however resprouters and fire plays an important role in the development of restiad dominated wetlands where the underground parts of *E. minus* are protected and survive fire (Moss *et al.* 2016). *E. minus* resprouts within weeks of a fire and recovers rapidly and can reach pre-fire levels within 2 years of burning and will continue to grow considerably for up to 17 years post fire (Wahren and Walsh 2000; Wagstaff and Clarkson 2009).

This study investigates possible vegetation thickening and encroachment, related to changes in fire regimes, at the Moon Point wetlands, Fraser Island. This analysis will be based on historical aerial photographs combined with contemporary field data, using Idrisi' TerrSet Land Change model a land change modeller to identify alterations of three major vegetation types: sclerophyll forest and woodlands (Forest), *Banksia* forest monocultures (*Banksia*) and restiad wetlands (Wetlands). Through quantified assessment of the aerial photographs using fine-gauged graticules vegetation types were manually assessed and counted to identify changes in vegetation density. Further vegetation polygons were created for the three vegetation types for analysis through the land change modeller to identify alterations in the three vegetation types, transition potential for landscape change and to predict alterations in the Moon Point community structure into the future.

5.2 Study Area

Fraser Island (Figure 5.1) is situated within subtropical coastal eastern Queensland (25.2167° S, 153.1333° E) and is approximately 124 km long and 20 km wide. The average annual rainfall is between 1300 to 1700 mm/year with mean temperatures ranging between 14 °C in winter and 29 °C in summer (Donders *et al.* 2006). The soils are composed of marine and aeolian sand deposits with few scattered bedrocks of basalt, with the central portion of the island being composed of complex sand dunes that are vegetated and with numerous lakes occurring in the depressions between the dunes (Gontz *et al.* 2015). Vegetation is composed of coastal scrub and sandy plains, *E. minus* wetlands, open heathlands, open dune forests, and open forests on leached soils, *Syncarpia* forests, notophyll vine forests, as well as wet and dry sclerophyll forests that are dominated by *Eucalyptus* and *Corymbia* species and to open 'Wallum' heathlands (Srivastava *et al.* 2013). The vegetation of Fraser Island is laterally zoned following the contours of successive parabolic dune systems that run parallel to the islands coastline (Donders *et al.* 2006) older dunes generally lying to the west, overlaid partly by progressively younger dunes to the east (Lennon 2012; Moss *et al.* 2016). Nutrient availability increases with the age of the sand and amount of detritus initially then declines due to leaching through a retrogressive successional process (Walker *et al.* 1981; Longmore and Heijnis, 1999; Donders *et al.* 2006). Zonation and succession of plant communities are dictated by salinity, water table, age and nutrient status of dune sands, exposure and frequency of fires, which creates an east to west sequence of vegetation (Lennon 2012; Moss *et al.* 2016).

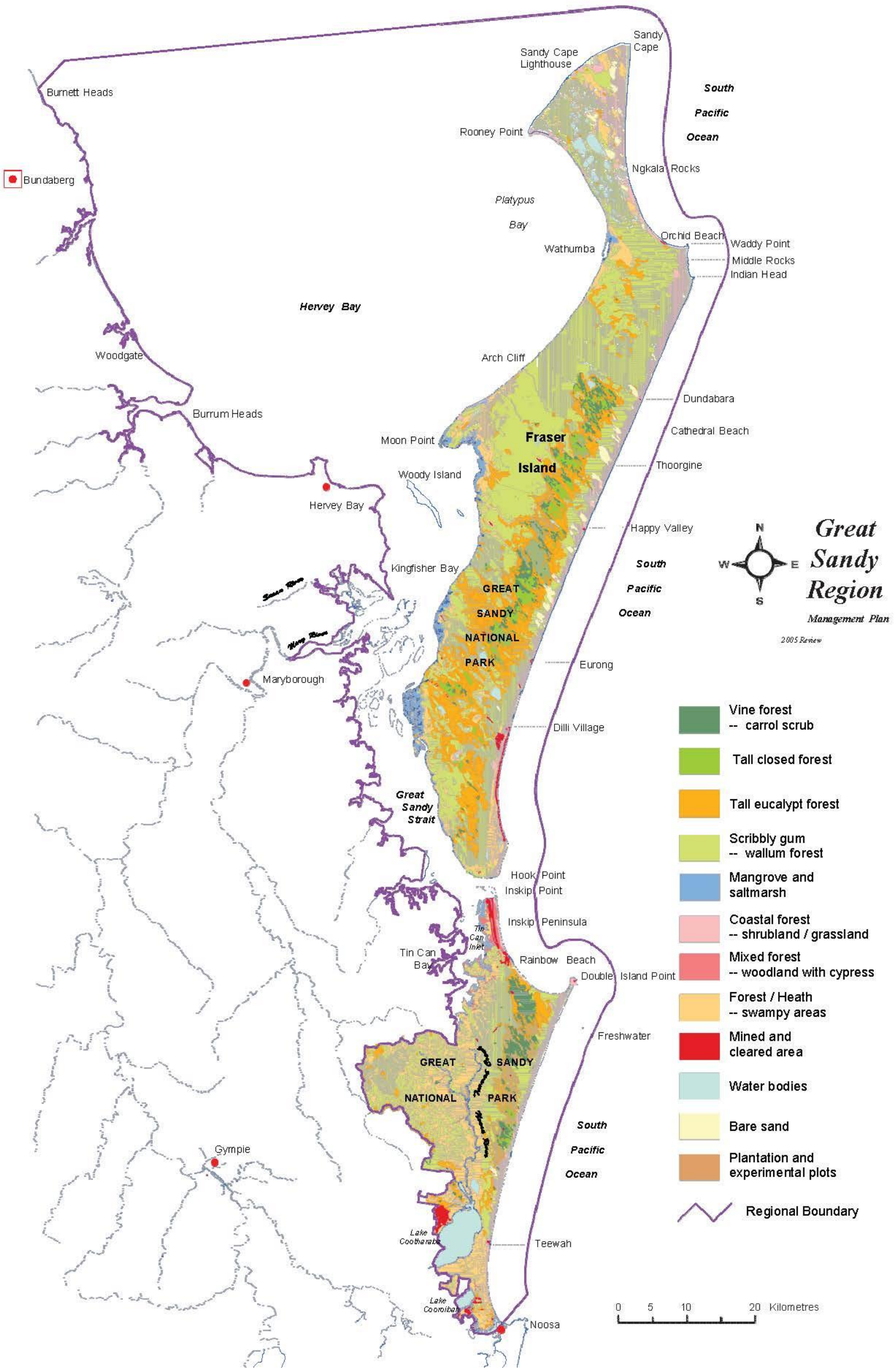


Figure 5.1: Fraser Island on the east coast of Queensland forms part of the Great Sand Region National Park which is World Heritage listed and the World's largest sand island. Source The State of Queensland. Great Sandy Region Management Plan 1994 -.

Moon Point (Figure 5.2) is situated on the most westerly coast of Fraser Island (25.1239^0 S, 153.0413^0 E) consists of mixed open sclerophyll forest/woodland, *E. minus* wetlands, mangrove communities, *Melaleuca* and *Banksia* forests, as well as salt marsh and dune vegetation. Common species of plants found within the ecotone (study area) between the *E. minus* wetlands and sclerophyll forests are listed in Table 5.1. The ecotone is the transition area between the *E. minus* and sclerophyllous forest ecosystems that occur along an ecological gradient showing the diversity of both boundary types (Odum and Odum 1953; Warman *et al.* 2013). The entire area is subjected to fire, which seems to play a role in reducing shrub cover Present fire management of Fraser Island is prescribed burning to promote biodiversity and reduce wildfire risk (Moss *et al.* 2016).

Table 5.1: List of common plant species found within the ecotone of Empodisma and sclerophyll vegetation periphery

Moon Point vegetation					
Genus	Specie	Common_name	Genus	Specie	Common_name
Acacia	*concurrens*	Black wattle	*Eucalyptus*	*tereticornis*	
Angophora	*leiocarpa*		*Eurychorda*	*complanata*	
Baeckea	*frutescens*		*Gahnia*	*sierberiana*	Red-fruited saw sedge
Baloskion	*tetraphyllum*	Tassel cord-rush	*Gahnia*	*clarkei*	Tall saw sedge
Banksia	*robur*	Swamp Banksia	*Gleichenia*	*dicarpa*	
Banksia	*serrata*	Serrated Banksia	*Gleichenia*	*mendellii*	
Banksia	*aemula*	Wallum Banksia	*Hakea*	*actites*	
Banksia	*oblongifolia*	Dwarf banksia	*Hibbertia*	*salicifolia*	
Blechnum	*indicum*		*Hypolaena*	*fastigiata*	
Boronia	*falcifolia*		*Lepironia*	*articulata*	
Callitris	*Columellaris*		*Leptospermum*	*liversidgei*	
Calytrix	*tetragona*	Fringe myrtle	*Leptospermum*	*neglectum*	
Cassytha	*glabella*	Dodder	*Leucopogon*	*ericoides*	
Cassytha	*pubescens*	Dodder	*Livistona*	*decora*	
Casuarina	*littoralis*		*Melalauca*	*quinquenervic*	
Caustis	*recurvata*		*Melalauca*	*dealbata*	
Conospermum	*taxifolium*		*Monotoca*	*scoparia*	
Corymbia	*intermedia*		*Ochrosperma*	*lineare*	Straggly haekea
Corymbia	*gummifera*		*Petalostigma*	*pubescens*	
Corymbia	*tessellaris*		*Petrophile*	*shirleyae*	Conesticks
Empodisma	*minus*	Wire rush	*Plachonia*	*careya*	
Epacris	*microphylla*	Coral heath	*Platysace*	*linearifolia*	
Epacris	*obtusifolia*	Common heath	*Pseudanthus*	*orientalis*	
Epacris	*pulchella*		*Ricinocarpos*	*pinifolius*	
Eucalyptus	*signata*	Scribbly Gum	*Sprengelia*	*sprengeliodes*	Lemon scented tea tree
Eucalyptus	*rasemosa*		*Strangea*	*linearis*	
Eucalyptus	*pilularis*		*Utricularia*	*uliginosa*	Bladder wort
Eucalyptus	*exserta*		*Xanthorrhoea*	*fulva*	Swamp grasstree

5.3 Methodology

Two aerial photographs of the same spatial area and extent with different temporal scales (1958 and 2010) for the Moon Point area were digitised and georeferenced (WGS 1984 UTM Zone 56S) using ESRI ArcMap. Fieldwork was undertaken in March 2016 for ground-truthing using a Garmin GPS to capture 42 waypoints to create polygons and to identify the present position of the three key vegetation types used in the land change analysis and consisting of Forest (surrounding sclerophyll and *Melaleuca* forest), *Banksia* (*Banksia* forest monoculture) and Wetland (predominantly *E. minus* dominated mire, including the patterned fens). The aerial photographs and ground truth data were then prepared for use in TerrSet's Land Change Modeller where a quantified assessment of vegetation change was undertaken. Collecting of ground truthing data was undertaken on the 4th to the 9th March 2016 tracing the boundaries with GPS.

5.3.1 Land Change Analysis Modeller

The two georeferenced aerial photographs of 1958 and 2010, along with the 2016 ground truth data were processed in ESRI ArcMap where the three broad vegetation types were created and added as polygon layers, merged, converted to a raster and then exported as ESRI image files. The three sets of data 1958, 2010 and 2016 were then imported into TerrSet and saved in Idrisi raster format for use in the land change modeller as land cover files. All three files prepared for use in TerrSet Land Change Modeler have the same spatial parameters and reference system. This includes the same projection, cell size, spatial extent, spatial resolution, rows and columns. Further the three land cover maps contained the same categories and sequential order. The land cover files were loaded into TerrSet LCM projects parameters panel for change analysis in date sequential order, with 1958 and 2010 first followed by 1958 and 2016 and then 2010 and 2016, and the underwent change analysis, with the spatial trend of change assessed to the 3rd order polynomial. Sub-models were grouped and run through the LCM transition potentials and finally through change prediction for future projections of vegetation alterations.

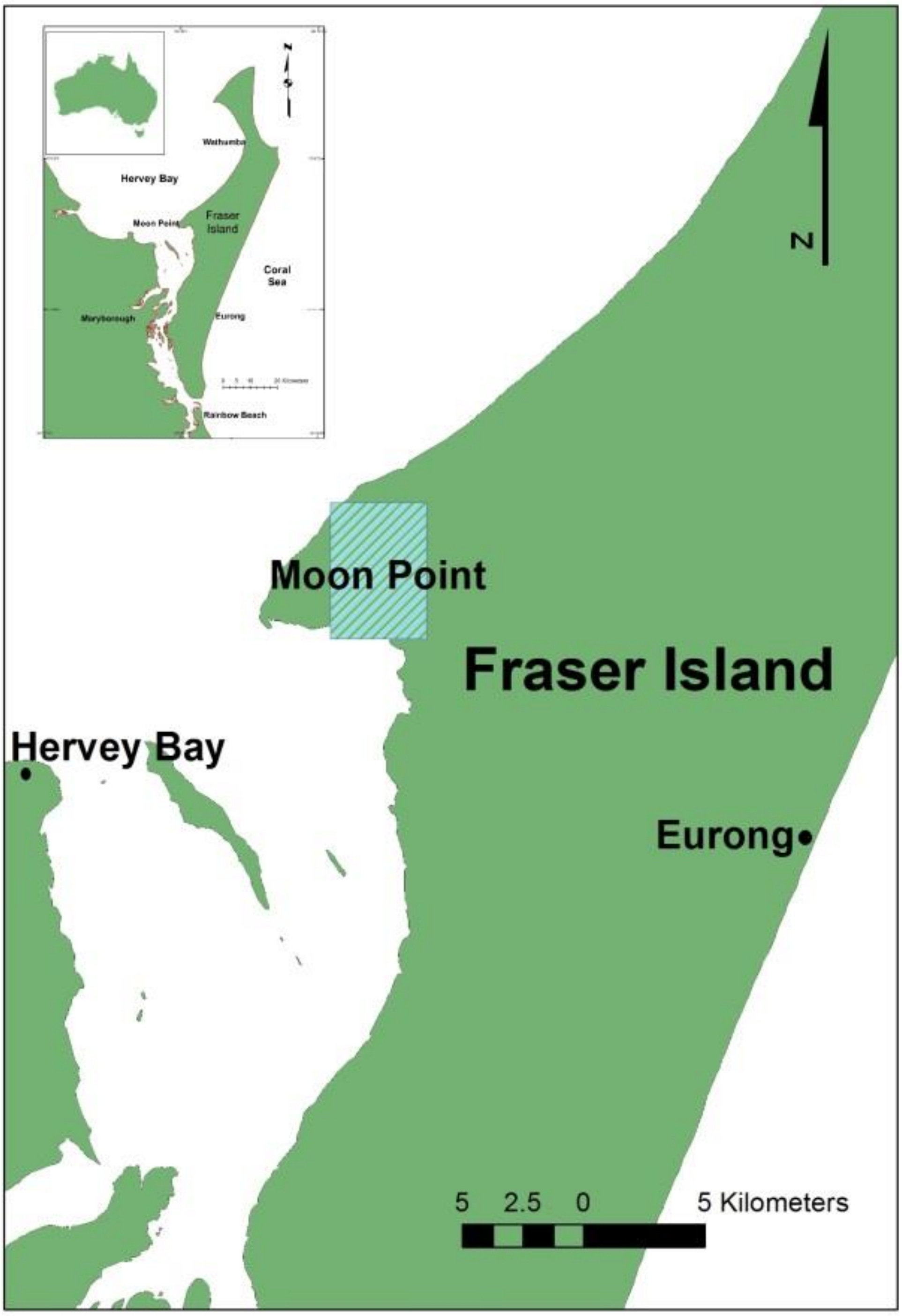
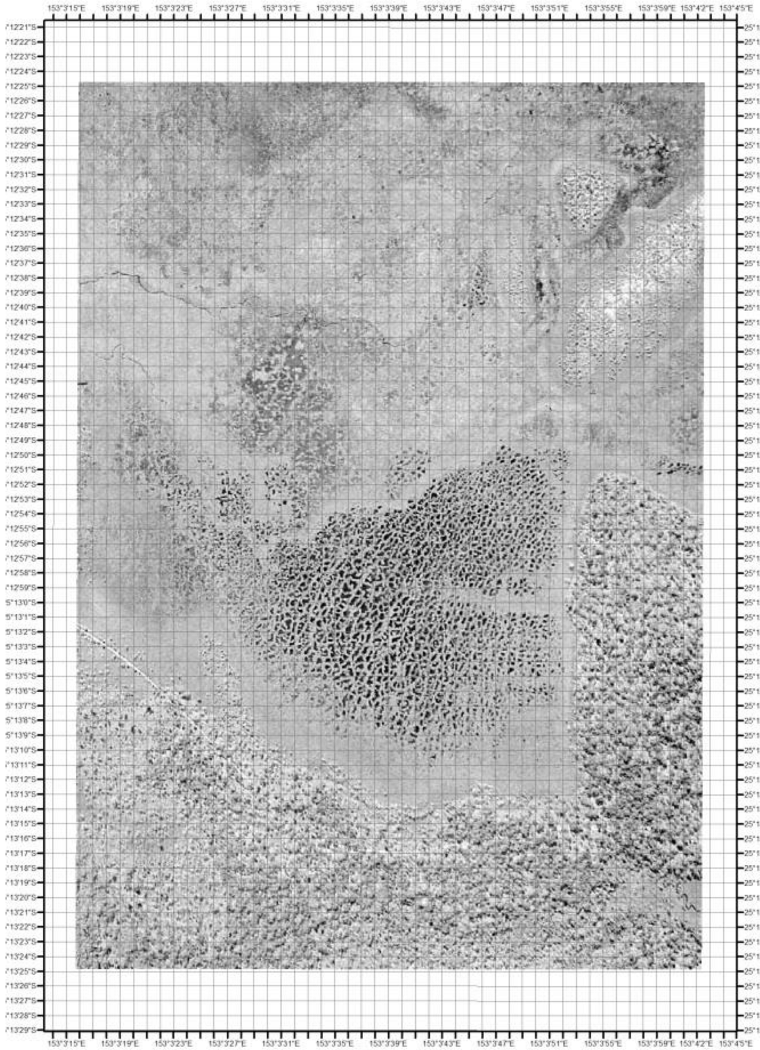


Figure 5.2: Moon Point is situated on the most westerly coast of Fraser Island

5.3.2 Quantified Vegetation Assessment

Grids and Graticules were created for the 1958 and 2010 Moon Point aerial photographs using meridians and parallels, placing parallels 1 second of latitude and meridians 1 second of longitude. At the equator an arc-second of longitude is approximately equal to an arc-second of latitude, which is 30.87 meters. Arc-seconds of latitude remain nearly constant, while arc-seconds of longitude decrease in a trigonometric cosine-based fashion as you move toward the Earth's pole. At -25.21 degrees south latitude, an arc-second of longitude equals 30.87 meters * 0.90476 (cos 25.21°) providing grids of 27.5 by 30.87 meters. The grids of the georeferenced 1958 and 2010 aerial photographs of Moon Point were used to identify Forest, *Banksia* and Wetland vegetation as well as change over time by summing all vegetation cover of each of the 3 classes that was >50% cover within each grid for each vegetation type (Figure 5.3).

a



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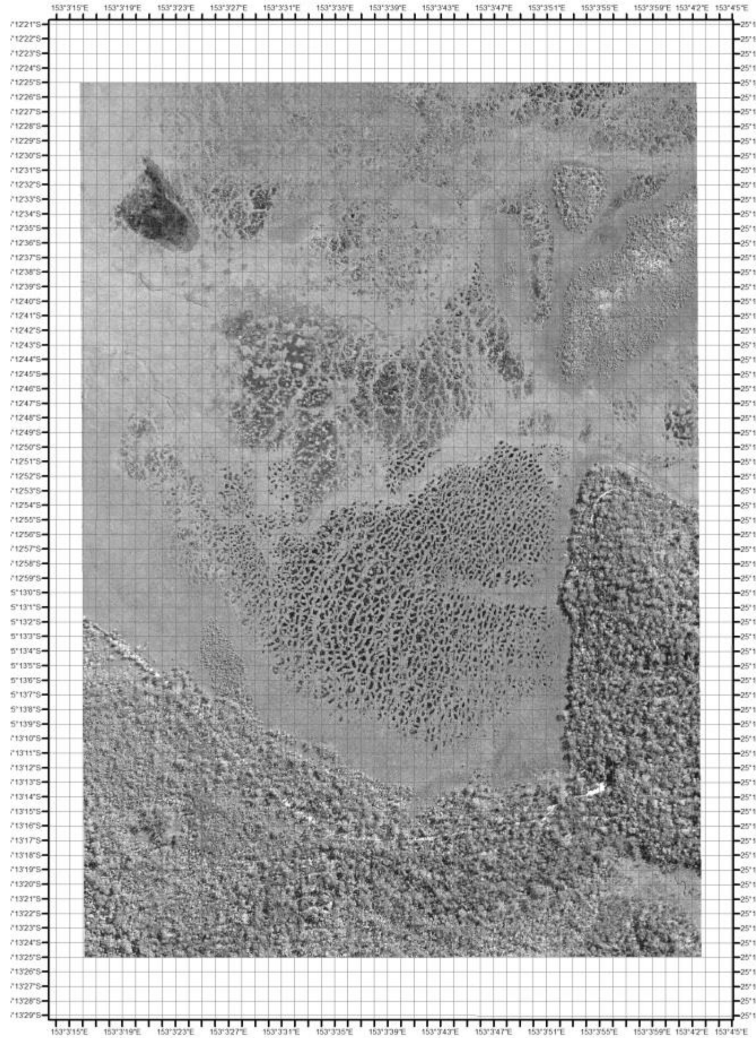

Figure 5.3. (a) 1958 and (b) 2010 aerial photographs of Moon Point, Fraser Island showing 1 second meridians and parallels or 27.5^2 meters.

5.4 Results

5.4.1 Land Change Analysis

In the first step the 1958/2010, 1958/2016, and 2010/2016 land cover maps were assessed for change between time 1 and time 2, with a focus on areas that transitioned from one land cover state to another. The dominant transitions are grouped and modelled and termed sub-models. These sub-models were combined and not run separately.

The second step in change prediction was transition potential modelling where the potential for land to transition was identified and transitional potential maps were produced, which were organised within an empirical evaluated transition sub-model with the same underlying driver variables. The variables were used to model historical change process. The transitions were modelled using the multi-layer perceptron (MLP) neural network and once calibrated were used to predict future scenarios.

The third step was change prediction where we used the historical rates of change and transition potential model in the LCM to predict future scenarios for the specified year 2066, fifty years into the future.

5.4.1.1 Step 1. Change Analysis

The land cover images (earlier and later land cover image) for (1) 1958 to 2010, (2) 2010 to 2016 and (3) 1958 to 2016 of Moon Point produced gains and losses by category where two units were selected for analysis of land change, (a) percentage change (per cent change) and (b) percentage of area (per cent of area) (Figure 5.4), where Percent Change = (number of pixels changed for a class / area of a class in the later land cover image)* 100 and Percent Area = (number of pixels changed for a class / total area of the land cover map)*100.

The analysis produced two Gant charts of gains and losses between Wetland, *Banksia* and *Forest* for the different aerial photograph time scales. The percentage (%) change (Figure 5.4, a), shows that *Banksia* had the greatest gains for this land cover category of 6.6% with some losses of -1.5%. Forest increased by 4.0% with only minor losses of less than -0.4%. Wetland overall had the greatest losses for the same period to > -2.0% with a minor gain of 0.2%. Between 2010 and 2016 the percentage (%) change shows an increase in *Banksia* to 12% with a 1% loss, Forest gaining to 3% however with a 0.1% loss and Wetland with a loss of -1.9% and a 0.1% gain. Between 1958 and 2016 the gains and losses are similar to 2010 and 2016 gains and losses, however with a decrease in losses for both Forest and *Banksia* compared to 2010 to 2016, with *Banksia* showing a loss of 0.1% and Forest almost zero, Wetlands showing the greatest loss of -3.8%.

The percentage of area gains and losses (Figure 5.4, b) show that overall Forest has gained substantially over Wetlands, approximately 1.24% for 1958 to 2010, a 1% gain between 2010 and 2016 and a 2.1% gain between 1958 and 2016. Wetlands having the greatest losses, approximately -1.45% for 1958 to 2010 and -3.8% between 1958 and 2016. With *Banksia* showing only minor gains of approximately 0.27 for 1958 to 2010 and 0.4% for 1958 to 2016.

Spatial trend of change to the 3rd order of polynomial were analysed between wetlands and forest and wetlands and *Banksia*. Each forest and *Banksia* were independently mapped. The spatial trend of change wetlands to all shows that wetlands have reduced in area and in a north to north westerly direction with forest increasing from the west, south, and east, with *Banksia* increasing from the east. The spatial trends of change for wetlands to forest and wetlands to *Banksia* confirm this shift (Figure 5.5).

5.4.1.2 Step 2. Transitional Potential Modelling

The transition potential grouped transitions between the two land cover maps into a set of sub-models where each sub-model was identified with a set of driver variables. The transitions were modelled using a multi-layer perceptron (MLP) neural network to predict future scenarios (Figure 5.6) for the land cover images 1958 to 2010, 2010 to 2016 and 1958 to 2016. Transition potential maps for each transition were derived with an expression of time-specific potential for change. The greatest potential for transition is seen in the land cover images 1958 to 2016 with Forest and *Banksia* showing possible extensive encroachment and thickening (Figure 5.6. 3, a and b).

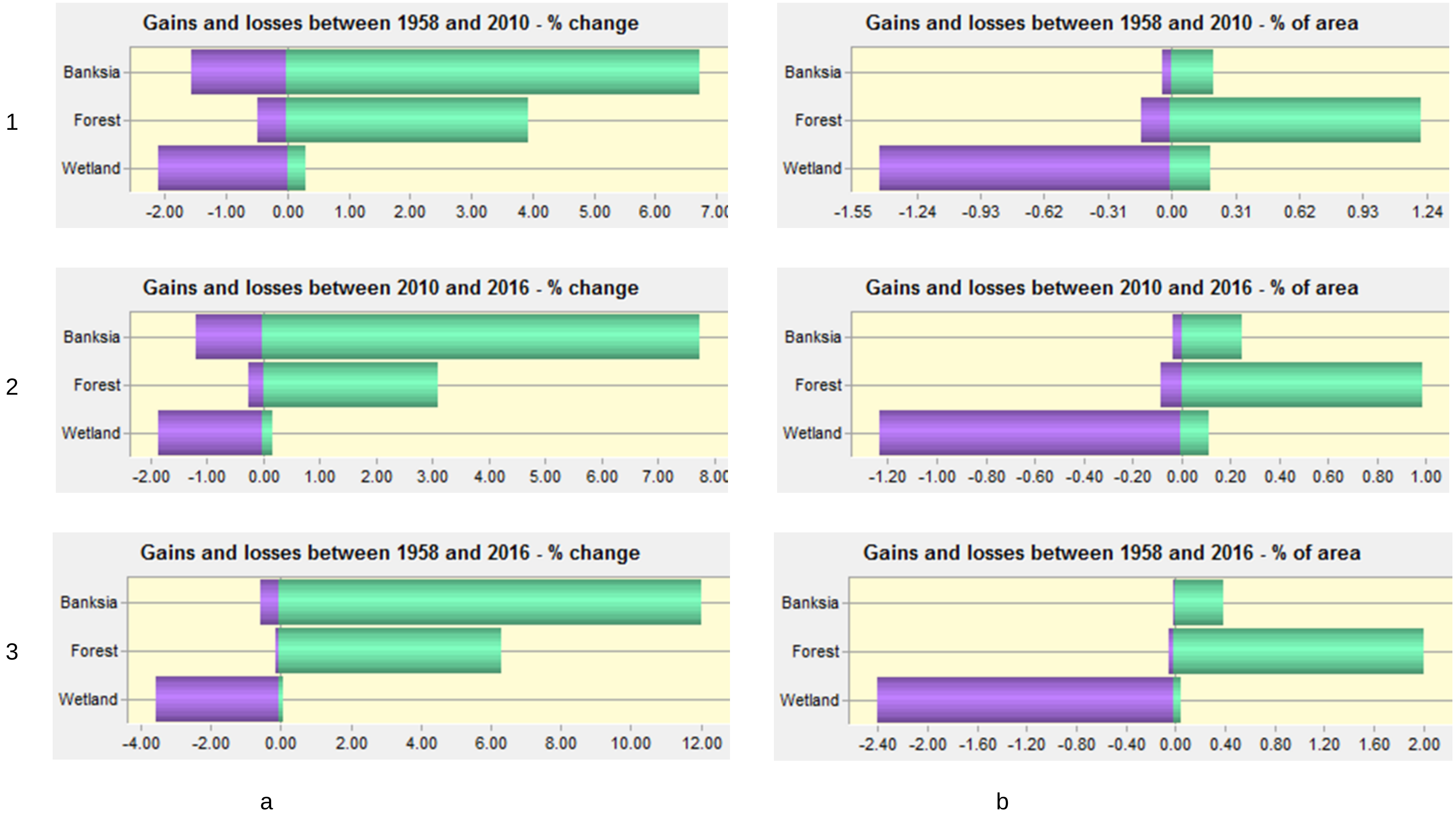

Figure 5.4. Gantt charts showing the gains and losses between wetlands, forest and banksia at Moon Point between (1) 1958 and 2010; (2) 2010 and 2016; (3) 1958 and 2016. Both (a) and (b) showing substantial losses for wetlands over this period. (b) Forest has gained the greatest % of area during this period.

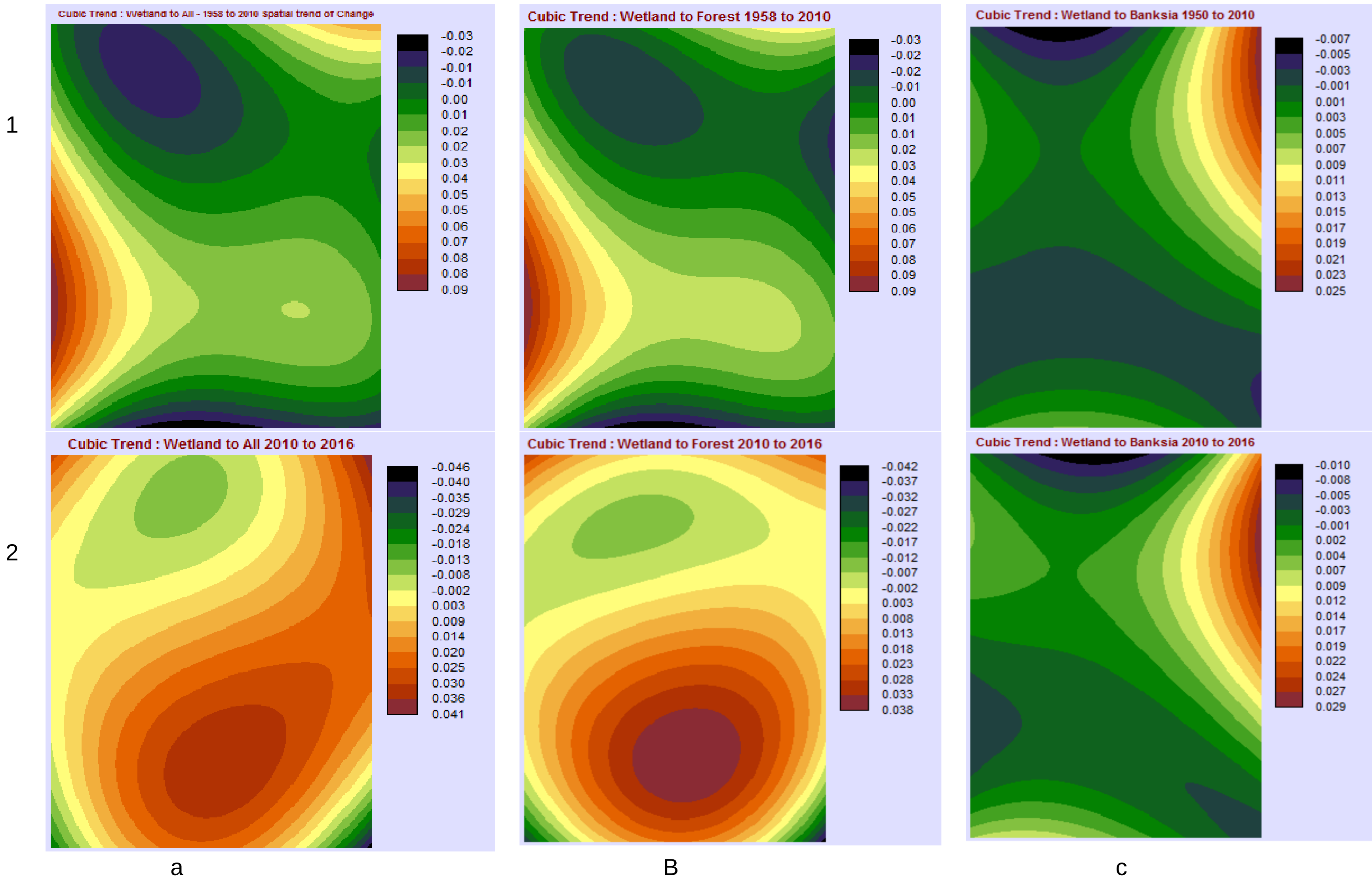

Figure 5.5: Spatial trend of change to the 3rd order polynomial for (1) 1958 to 2010, (2) 2010 to 2016 and (3) 1958 to 2016. Positive values reflect transition from (a) wetlands to all (b) wetlands to forest and (c) wetlands to banksia.

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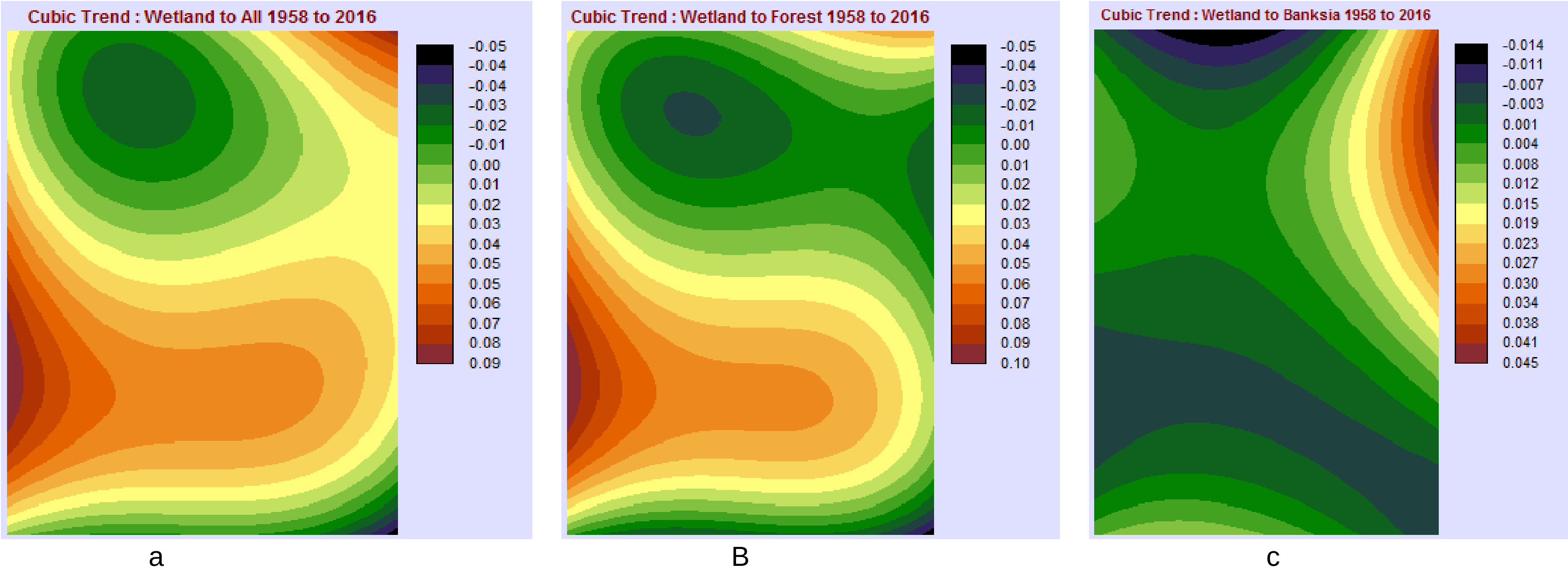


Figure 5.5: Continued - Spatial trend of change to the 3rd order polynomial for (1) 1958 to 2010, (2) 2010 to 2016 and (3) 1958 to 2016. Positive values reflect transition from wetland to (a) wetlands to all (b) wetlands to forest and (c) wetlands to banksia.

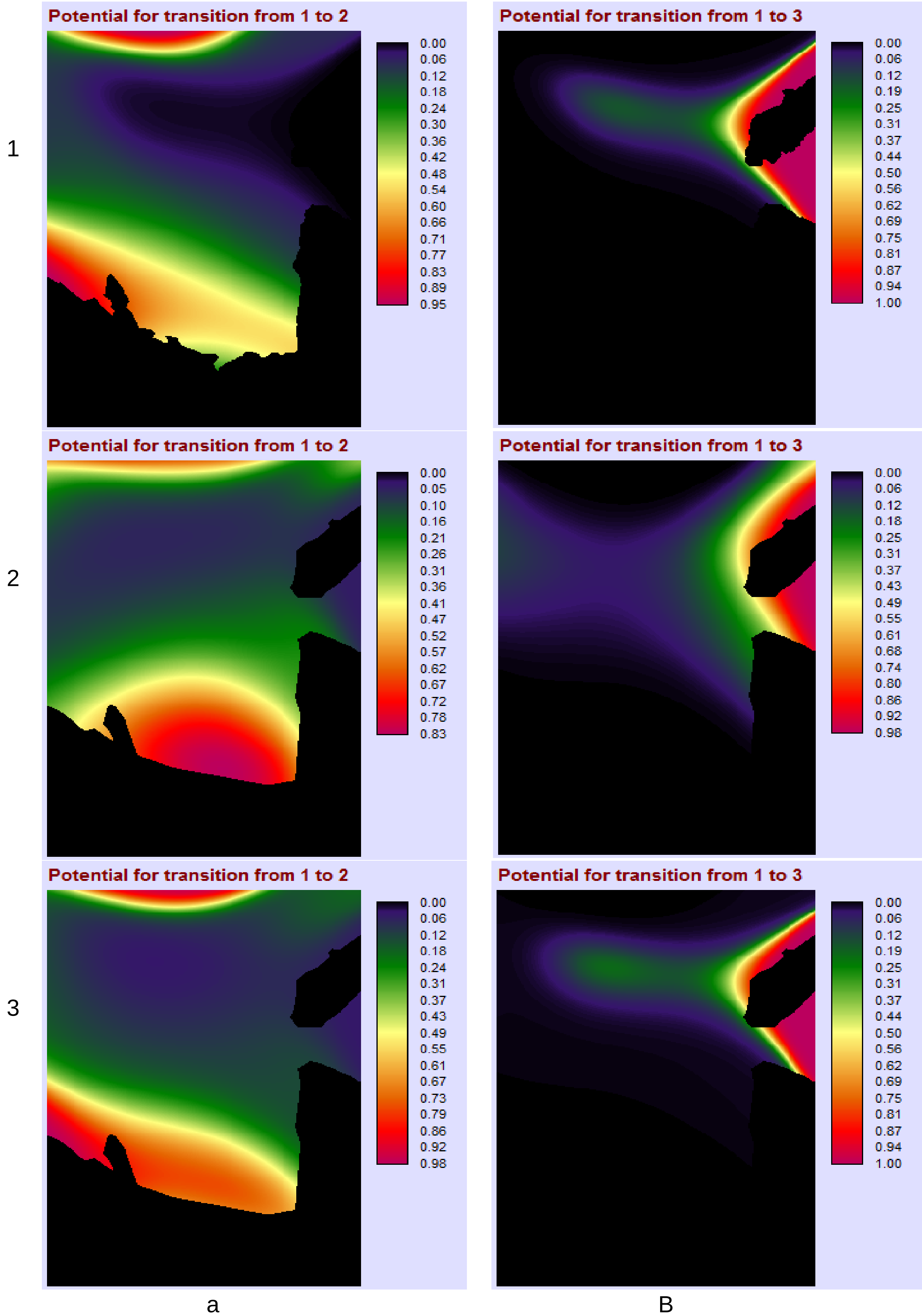


Figure 5.6. Potential for transition from (a) 1 to 2 and (b) 1 to 3. (a) showing the potential for wetlands to transition to forest and (b) the potential for transition from wetlands to banksia. (1) 1958 to 2010, (2) 2010 to 2016 and (3) 1958 to 2016 showing potential for transformation.

5.4.1.3 Step 3. Change Prediction

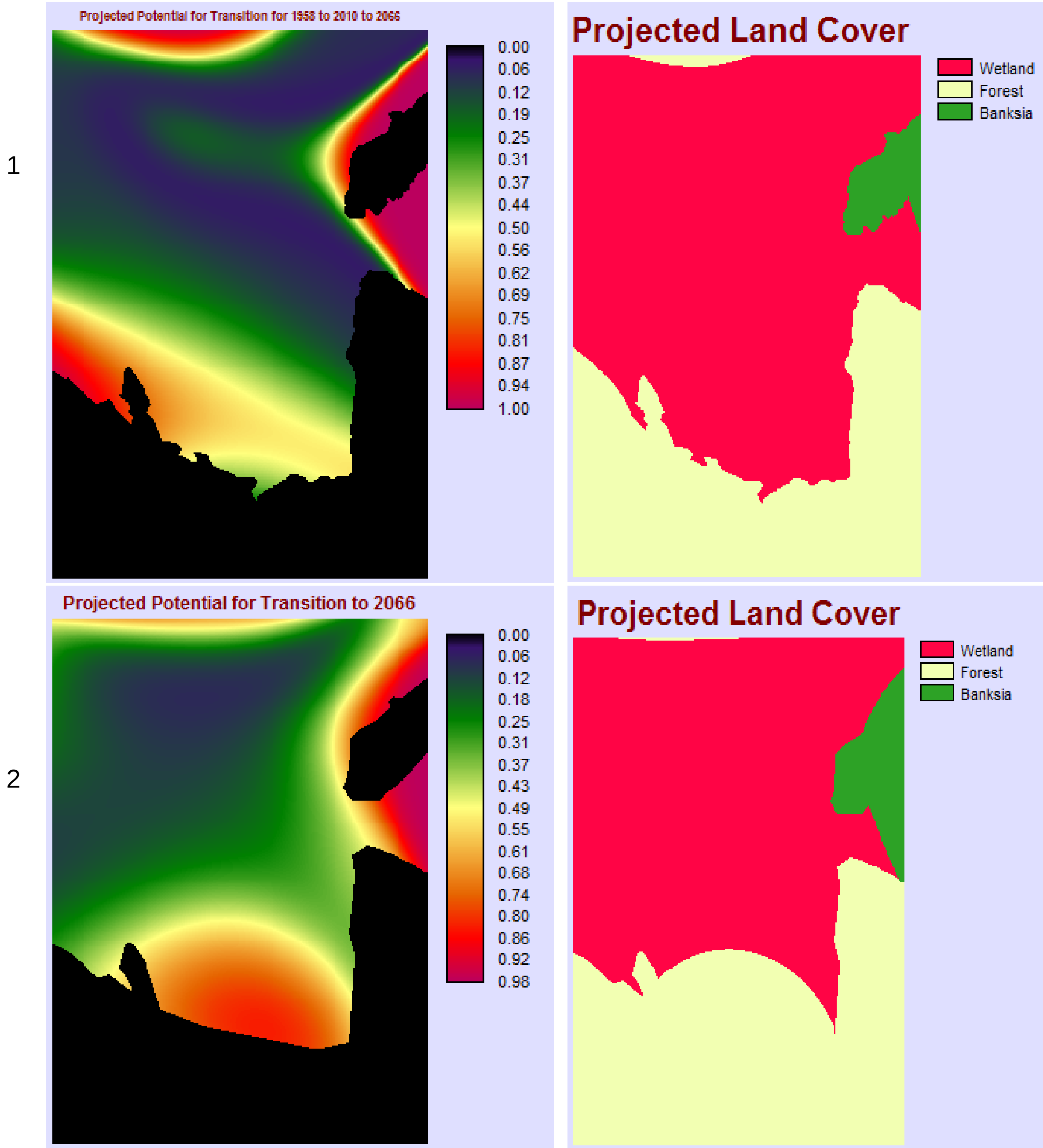
The LCM used the transition potential model and historical rates of change to predict a future scenario projected to 2066 (Figure 5.7). The model determined how variables influenced future change and how change took place between time 1 and time 2, calculating transition to 2066. A soft mode of change prediction was chosen as the output is a continuous mapping of vulnerability to change. According to Eastman (2015) soft prediction output is a continues mapping of vulnerability to change, giving the degree to which the areas have the right conditions to precipitate change, compared to hard prediction that is a commitment to a specific scenario.

The results do not state it will occur, however the degree to which the area has the right conditions to precipitate change. It used an aggregate of the transition potentials of all selected transitions.

5.4.2 Quantified Vegetation Assessment

Total counts of fine-gauged graticule grids for each vegetation type (Forest, Wetland, *Banksia*) for both the Moon Point georeferenced 1958 and 2010 aerial photographs (Figure 5.3) showing gains and losses, with the greatest change for wetlands with a loss of -2.07% or -58 grid loss between 1958 to 2010, banksia showed a 0.53% gain or 15 grid increase between 1958 and 2010 and forest with a 1.53% gain or 43 grid increase for the same time period (Table 5.1) % of area.

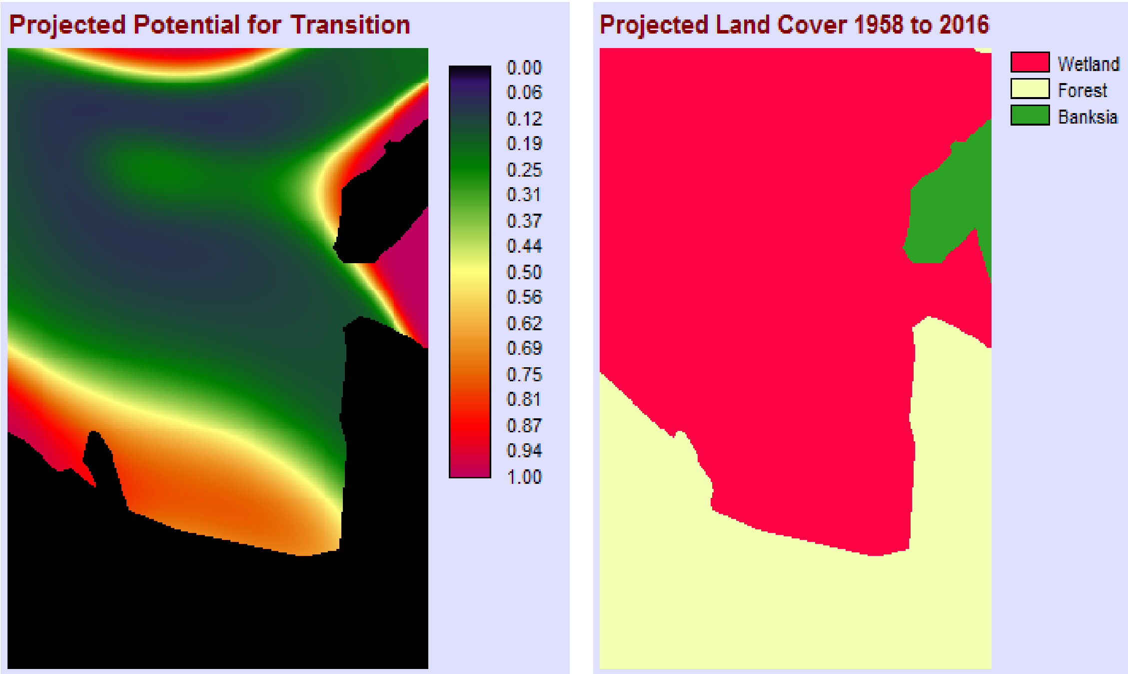
Comparative analysis of the data taken from quantified analysis and the LCM show interesting similarities in both gains and losses between forest, banksia and wetlands. With the manual count % change giving a -2.07% loss of wetlands with % area change and % change for wetlands showing -1.45% and -1.90% respectively. Both Forest and *Banksia* data showing gains (Table 2) over the time period 1958 to 2010 (52 years).



a (soft prediction) b (hard prediction)

Figure 5.7. Change prediction projected to 2066 for (1) 1958 to 2010, (2) 2010 to 2016 and (3) 1958 to 2016. (a) Soft prediction showing vulnerability to change and the degree to which the area has the right conditions to precipitate change. (b) Hard prediction showing change from wetlands to forest and banksia at 2066.

3



a (soft prediction) b (hard prediction)

Figure 5.7: Continued. Change prediction projected to 2066 for (1) 1958 to 2010, (2) 2010 to 2016 and (3) 1958 to 2016. (a) Soft prediction showing vulnerability to change and the degree to which the area has the right conditions to precipitate change. (b) Hard prediction showing change from wetlands to forest and banksia at 2066.

5.5 Discussion

5.5.1. Land Change Analysis

Quantified vegetation analysis has revealed changes in vegetation thickening over the Moon Point landscape for a 52 year period based on the 1958 and 2010 aerial photographs. With the vegetation thickening there has been a corresponding loss of the *E. minus* wetlands as vegetation encroaches onto this area. The change in vegetation dynamics and composition show a positive trend over the past 58 years to present (based on the ground truth data collected in 2016) with the Wetland showing losses (Table 5.2), while both Forest and *Banksia* displaying gains over this time (Table 5.3). The LCM show similar results to the manual grid count taken from the two aerial photographs, with losses to Wetlands and gains to both Forest and *Banksia*, however the LCM analysis provided analysis for cubic trend for transformation, potential for transition and projected land cover potential for transition to 2066. Further, additional data was used in the analysis where the ground-truthed data for March 2016 was analysed and compared to the 1958 and 2010 aerial photograph data sets.

Potential for transition supports the cubic trends, however provides for prediction of 50 years into the future to 2066 using trends from 1958 to 2010, 2010 to 2016 and 1958 to 2016. Projected land cover potential for transition to 2066 show that under the present status-quo Forest and *Banksia* will become more prominent in the landscape with a loss of *E. minus* wetland, which will continue to show losses, decreasing in area substantially over the next 50 years to an expected 30% reduction in current extent under present trends (Figure 5.7). However there are limits to the actual expansion of forest to wetlands and *Banksia* to wetlands as the wetlands cannot support forest and *Banksia* except on the margins.

Table 5.2. Results of a manual count of grids between the 1958 and 2010 georeferenced aerial photographs using parallels of 1 second of latitude and meridians of 1 second of longitude.

Manual-count Land Change of Moon Point							
Counts				**Percentage Change**			
	1958	**2010**	**Change**		**1958**	**2010**	**% of area**
Forest	867	910	43	**Forest**	30.90%	32.43%	1.53%
Banksia	74	89	15	**Banksia**	2.64%	3.17%	0.53%
Wetland	1865	1807	-58	**Wetland**	66.46%	64.40%	-2.07%
Total	2806	2806					

Table 5.3 Results for the analysis of the 1958 to 2010 aerial photographs over 52 years of % of area change, manual count and per cent change'. Per cent of area change' and the manual count per cent change' show similar change trends, where per cent change' shows overall percentage change that has taken place over the 52 year period for each vegetation type.

% of Area change			Manual Count % change		% Change	
Forest	1.24%	Gain	1.55%	Gain	4.0%	Gain
Wetland	-1.45%	Loss	-2.07%	Loss	-1.9%	Loss
Banksia	0.77%	Gain	0.53%	Gain	6.6%	Gain

Vegetation thickening, the key process that is associated with the reduction of the *E. minus* wetlands, may be associated with changes in fire regime (Moss *et al.* 2016) with Figure 5.8 display a fire frequency of approximately every 12 years for the Moon Point region. Fires have been both wildfires and prescribed burns within the area of investigation, with four wildfires one in 1969, with the second fire in 1982, third in 1994 and the fourth in 2006 (Figure 5.9). A prescribed burn of the area was undertaken in June 2016 (pers.com Linda Behrendorff 24th June 2016). From this it can be seen that fire frequency has been approximately a fire every 11 to 12 years.

5.5.2. Vegetation Thickening

Changes in the fire frequency and lengthening the interval between fires has been shown to rapidly promote the increase of woody species in grasslands near forests (Swaine *et al.* 1992;) increasing vegetation cover (i.e. thickening) that may have implications for vegetation structure and function into the future (Pricope *et al.* 2015). Further, it has been shown that prescribed fire and fire return intervals of greater than 10 years can result in transition from open vegetation communities to woodlands within 30 to 50 years (Ratajczak *et al.* 2016). However, Stevens *et al.* (2016) suggests that woody encroachment may be due to changes in atmospheric CO_2, changes in land management and rainfall and not necessarily changes in fire frequency. Kgope *et al.* (2010), has shown that in experimental studies elevated CO_2 levels can help saplings escape fire by increasing growth rates so that the tree height is above the fire threat, which is also assisted by increasing the saplings underground carbon store (Saintilan and Rogers 2015). Kelley and Harrison (2014) state that with vegetation, fire interactions and CO^2 fertilization in wooded areas in Australia, that an increase in fire promotes a shift to more fire adapted trees with an overall increase in forested area, which increases carbon stocks. Further according to Harrison and Kelley (2016), CO_2 effects were found to increase burnt area in arid regions while increasing vegetation density and reducing burnt areas in forested regions, however CO_2 fertilization will be limited when taking nutrient limitation into consideration (Kelley and Harrison 2014). Further, Fensham *et al.* (2005), states that in semi-arid regions of Australia fire and woody encroachment showed no relationship at all. However, according to Baker and Catterall's (2015) study of Byron Shire vegetation dynamics, changes in vegetation structure following fire exclusion has resulted in the displacement of treeless ecosystems by forest, as well as the transition of open forest to rainforest. Regular anthropogenic fires by Aborigines prior to the arrival of Europeans would have prevented vegetation thickening, however with the settlement of Europeans in Australia and their fire exclusion or prevention techniques would likely have led to changes in vegetation and possible thickening (Gifford and Howden 2001). There is evidence that European fire management has altered the vegetation composition, with an increase in eucalypts relative to Casuarinaceae on Fraser Island and North Stradbroke Island (Moss *et al*. 2011; Moss 2013; Moss *et al*. 2016).

According to Moss *et al*. (2016) vegetation thickening is possibly a longer-term threat to the *E. minus* mires by the increase in myrtaceous arboreal taxa as shown in the post-European settlement sections of the Moon Point record. Further a major cause of vegetation thickening is the result of moving away from traditional fire management practices in the region (Russell-Smith *et al*. 2003; Moss *et al.* 2016). Murphy *et al.* (2015), states that any factor that decreases fire frequency is likely to increase the chance of forest formation, factors being for example, rockiness, topographic fire protection, insularity. Vegetation thickening is evident in sand environments through the accumulation of woody stems in fire-free years with midstorey trees showing some evidence of structural suppression in response to frequent fires (Vigilante and Bowman 2004). Gill (1975) describes fire frequency as a function of the number of fires experienced by a plant community within a given time period and that fire is a normal environmental variable. Research by Sheuyange and Weladji (2005) shows that vegetation responds differently to fire frequency, confirming the important role anthropogenic fires play in ecosystems. According to Spence and Baxter (2006) habitat structure was strongly influenced by fire frequency.

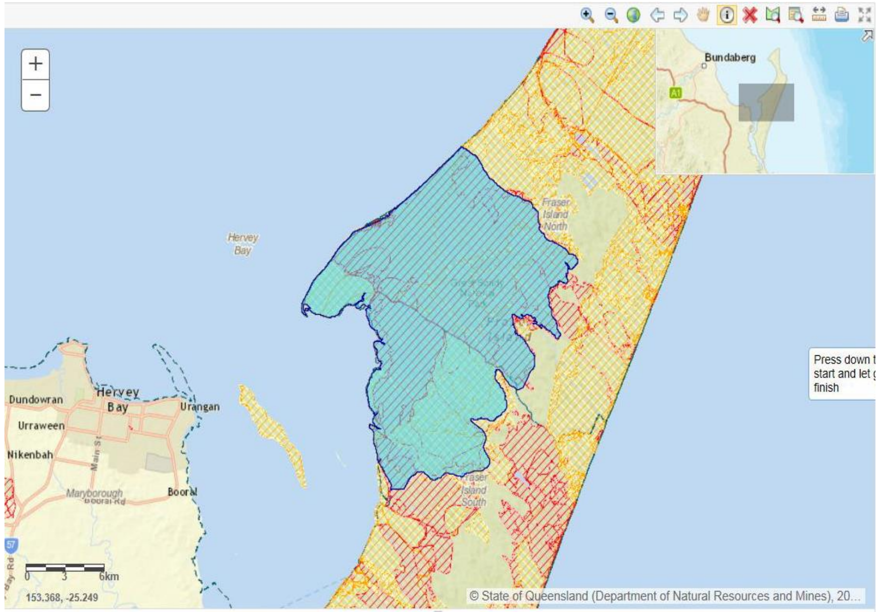
5.5.3. Implications for Management

Micro and macro charcoal and pollen records show that fire has played an important role at Moon Point with records from 25,000 years ago to present recorded in the peat records (see Chapter 4). Changes in climate and fire regimes have seen a change in vegetation composition and structure of Moon Point from rainforest taxa of predominantly Cunoniaceae, *Elaeocarpus*, Myrsinaceae and *Argyrodendron* to more sclerophyll arboreal taxa with an increase in rush abundances. Both Chapter 4 and Moss *et al*.(2016), reveal a significant change in fire regimes since European arrival and settlement which appears to be related to the increased presence of arboreal taxa supporting this land-cover change analysis (Farrell 2016).

Fire is an important disturbance regime of Fraser Island and its role in maintaining and altering forest vegetation is evident in the palaeoecological records of the Island, which, is also recognised by managers as an important driver of ecological change (Whitlock *et al.* 2003). If present conditions continue it is expected that further encroachment onto the *E. minus* wetlands will continue as projected by the LCM forecast for the year 2066.

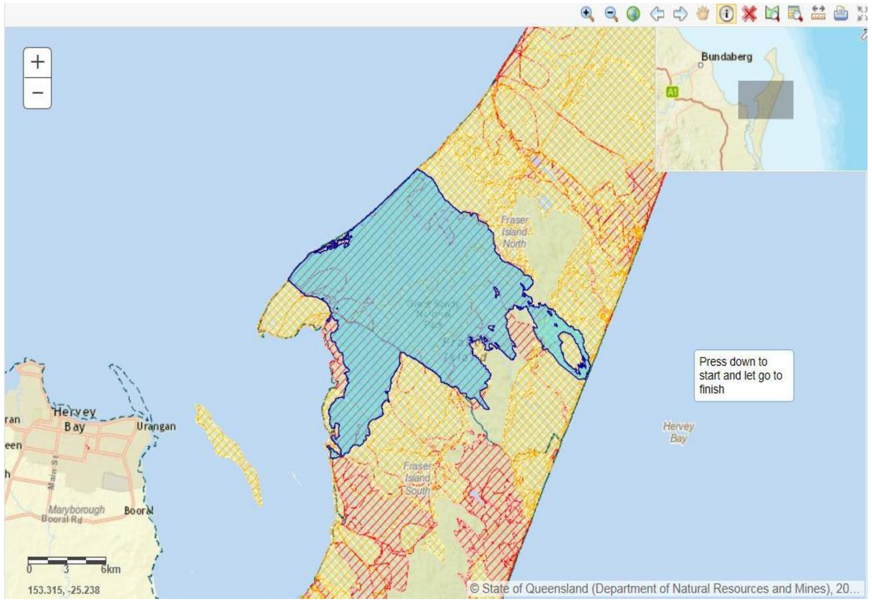
The *E. minus* wetland is under threat from changes to fire regimes, particularly in a higher CO_2 world that will include the threat of higher sea-levels and sea-water intrusion into the wetlands requiring important decisions to be made about fire management at Moon Point and other *E. minus* dominated sites. It is suggested that a fire frequency of 7 to 15 years with variation in season be implemented, avoiding set seasonal burns as this can favour particular species within the community over time. Based on the palaeo studies under the right circumstances *E. minus* wetlands are highly resilient and will respond to dramatic environmental changes such as the recovery from the mid-Holocene high-stand event (Moss *et al.* 2016).

(a)



January 1994 wildfire

(b)



November 2006 wildfire

Figure 5.8. Maps showing area burnt by wildfires at a frequency of 12 years. (a) Wildfire in January 1994 that burnt an area of 25728.41 ha. (b) Wildfire in November 2006 that burnt an area of 18917.21 ha. Source: Queensland State Government.

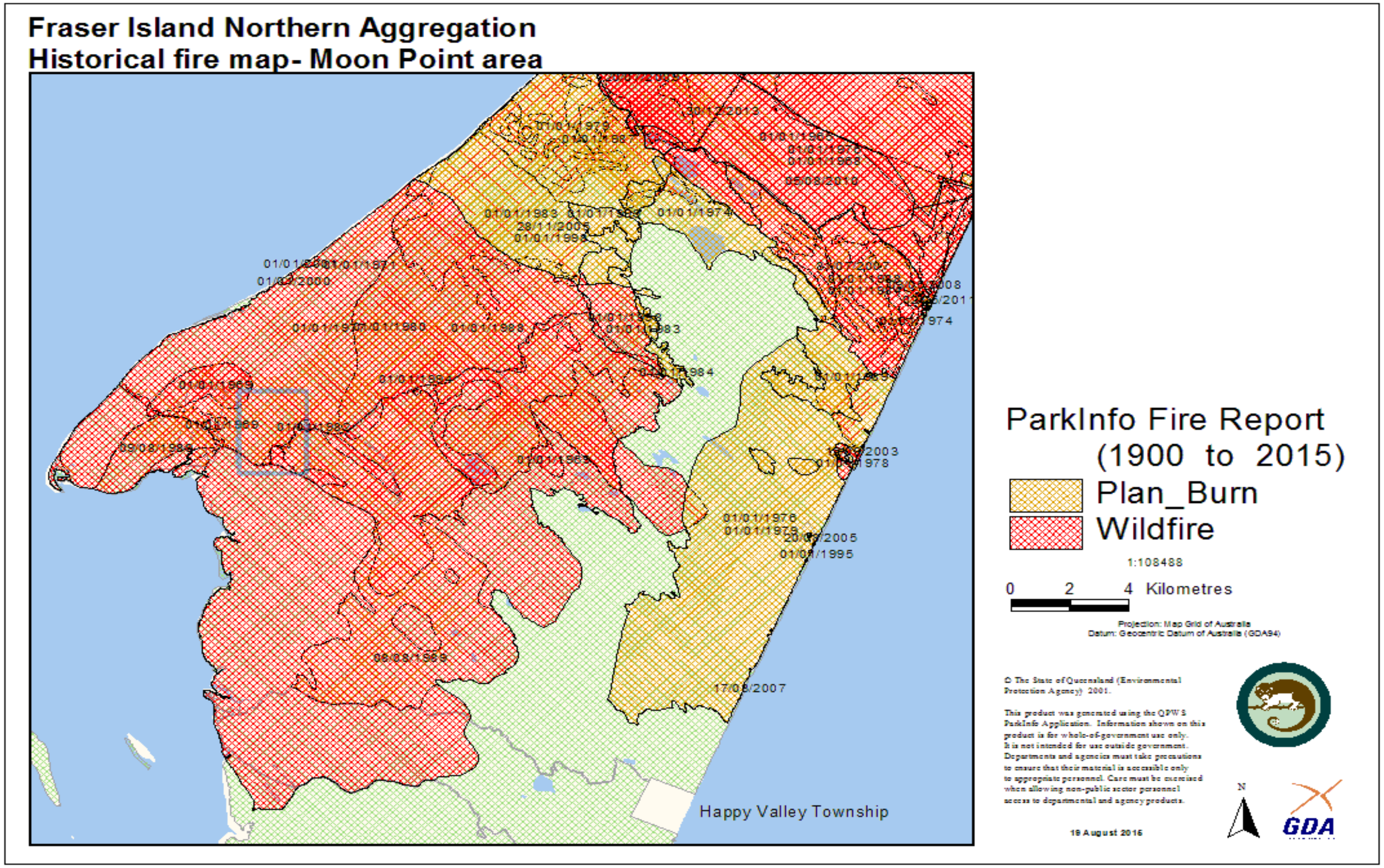


Figure 5.9: Fraser Island fire history from 1900 to 2015 showing the location of the Moon Point vegetation analysis area (grey rectangle centre left). Wildfires occurred in 1969, 1982, 1994 and 2006 Source Parks and Wildlife Services, Queensland State Government.

5.6 Conclusion

There is evidence that the wetlands of Moon Point are changing in the vegetation structure and composition with a decrease in *E. minus* wetland area as vegetation thickening and encroachment is taking place around the periphery of the Moon Point site. Future projection of thickening and encroachment of forest to the year 2066 according to the LCM shows gains over the wetlands between 1958 to present with a prediction to have the highest thickening impact over wetlands to 2066. Between the period 1958 to 2010 there have been limited fires of approximately one every 12 years. In April/May 2016 a prescribed burn of the area was undertaken which was in the cooler months of the year resulting in a cooler burn-off. Fire regimes appear to be the key factor with vegetation thickening at Moon Point, although an increase in CO_2 and changes in temperature and precipitation may also influence the rate of forest encroachment however further studies are needed. Change in fire management is probably the most cost effective adaptation strategy available in combination with continued conservation.

The *E.minus* wetland, sclerophyll forest ecotone is dynamic where fire acts as a destabiliser of the vegetation community. Low intensity frequent fires prevent encroachment and thickening by sclerophyll forest, maintaining a fire induced semi quasi climax state. However the invasion by sclerophyll forest is limited away from dry areas. According to the Queensland Government (2017) regional ecosystem details for 12.2.12a fires are recommended every 8 to 20 years with a burn mosaic of 40 to 80%.

Chapter 6. Fire, Climate Change and the FireBGCv2 Landscape Fire Succession Model.

Abstract

The aim of this study is to identify the use and application of a spatially explicit fire model within an Australian setting with a focus on Fraser Island within the subtropical region of eastern Australia. The fire simulation model FireBGCv2 landscape fire succession model is a research simulation platform for exploring fire and vegetation dynamics that has five inherent scales built into its design that correspond to spatial domains or organizational layers The model has two main types of files, parameter files and input files. It is a C++ computer program, which is intended as a research tool that can be used as a predictor of change. Managers need tools such as models to predict how changes will affect landscape composition and ecosystem function. Further models need to simplify real world processes such as the FireBGCv2 model that simplifies or excludes some biophysical processes affecting disturbances. However one important limitation of FireBGCv2 fire simulation model is the lack of a linkage between fire spread and fire behaviour with fuels and weather.

6.1 Introduction

This chapter examines fire modelling, the advantages and disadvantages and their use as a tool to provide managers of Fraser Island and the Great Sandy Region (GSR) with the ability to assess the possible impacts and changes to ecosystems by both wildfire and prescribed burning. Land managers require tools such as spatially explicit models to assist in their decision making processes as managing natural systems is complex and dynamic, however spatially explicit models need to be user friendly (Turner *et al*. 1995). According to Keane *et al.* (2001) biophysical models require abundant data, are complex and difficult to understand and require extensive field sampling to construct accurate maps and expertise in fire and fuels modelling. However one advantage is that the method is reproducible and can be updated with new data (Auerbach 2015). In this chapter the FireBGCv2 fire simulation model is assessed as a management tool, however due to the complexity of the model it was not possible to initialise and execute model runs except for simulated landscapes.

Wardell-Johnson *et al.* (2015) states that interactive effects of climate change and other threatening processes will have greater impacts on Fraser Island than climate change alone, particularly where climate change interacts with invasive species and with disturbances such as fire to transform ecosystems (see chapter 3). An increase in the duration and severity of the fire season due to the decrease in moisture and increase in temperature are likely to have a significant impact on biodiversity (Wardell-Johnston *et al.* 2015). There has been considerable speculation about the sensitivity of fire regimes to climate change. However, most of the discussion to date has not involved rigorous mechanistic analysis (Cary *et al.* 2002), with the suggestion that fire impacts (Cary 2002) on Fraser Island will increase under projected climate change, but little attempt to quantify the effects on vegetation structure and pattern.

Climate is a major driver of fire regimes in many ecosystems where climatic variation interacts with fire over multiple temporal scales (Bowman *et al.* 2009; Liu and Wimberly 2016). In fire prone regions a major driver of ecosystem change is increased frequency and intensity of fire (Wilkin *et al.* 2016) events caused by a warmer, dryer climate (Colloff *et al.* 2016). Fire has been evident in Australia from charcoal records since the Paleogene (65 million years ago) and throughout the Quaternary period to present (Cary *et al.* 2012), and has been a consistent feature throughout most of the Australian landscape, increasing in importance with the evolution of sclerophyll vegetation (Lynch *et al.* 2007). There is evidence of anthropogenic fires dated to around 45,000 years ago through Aboriginal occupation and fire stick practises (Turney *et* al. 2001; Moss and Kershaw. 2007; Moss 2008; Moss *et al.* 2013; Jurskis 2015) in fire-prone vegetation. Fire is a major perturbation in fire prone ecosystems that modifies habitats and drives evolutionary change (Bowman *et al.* 2012). With predicted rapid climate change over the next several decades there is the potential for severe wildfires and increases in fire hazards due to longer, dryer and warmer fire seasons (Keane *et al.* 2000; Turco *et al.* 2016). As fire is a complex process involving interactions and feedbacks between biological, ecological and physical processes across multiple spatial and temporal scales (Harris *et al.* 2016; Bowman *et al.* 2016) it limits the ability for models to project future fire activities due to a changing climate (Harris *et al.* 2016).

Wildfires have increased in size, intensity and severity over recent decades and frequent low intensity surface fires in forest ecosystems have transitioned to severe high intensity fire as a result of climate change and modification of the landscape (Stewart and Moss 2015). These recent increases in large fires globally have caused concern about the influence that climate change and humans may have on future fire regimes (Pechony and Shindell 2010). According to Keeley and Syphard (2016), fire response to climatic variation is shown to be correlated to global circulation patterns such as the Pacific Decadal Oscillation or El Niño-Southern Oscillation, and that climate change is expected to aggravate fire impacts on natural ecosystems. Any prediction of future fire regimes requires an understanding of how temperature and precipitation interact in controlling fire activity (Keeley and Syphard 2016). Boer *et al.* (2016), states that most studies on the impacts of climate change on fire regimes do not take into account transformational changes that take place in the relationship between climate, fire and vegetation. Further, climate induced shifts in vegetation and in fire regimes can potentially change vegetation and fire related adaptations with detrimental effects on species survival and ecosystem function (Bowman *et al.* 2011; Flatley and Fulé 2016). Identifying general pathways through which climate change may alter fire regimes is a critical next step for understanding fire under a changing climate (Hessl 2011; Harvey 2016).

Through the use of projections based on climate modelling, Australia has been predicted to become increasingly warmer with more variability in precipitation over most of the continent during the 21st century, increasing the risk of wildfires (Harrison and Kelley 2016) and changing fire regimes. A number of fire regime characteristics have been well studied in the context of climate change whereas fire severity has received less attention (Parks *et al.* 2016). Added to this fact is that what fire severity research there is, has relied on correlation between field measurements of fire effects and simple spectral reflectance indices that do not measure heat output or plant physiological changes (Smith *et al.* 2016) and has rarely been derived from multi-decadal remote sensing datasets (Parker *et al.* 2016). Modelling and databased methods to understand fire and fire regimes have rapidly coevolved over the past few decades, with fire modelling based primarily on present day satellite and field data. Contextual information about contemporary fire regimes has been provided from historical analysis of charcoal from sediment (i.e. databased methods) (Hantson *et al.* 2016).

Currently, however, very few initiatives link these important data types to fire in the earth system. Therefore integration between paleoecology and ecological modeling is needed to understand climate-vegetation-human-fire linkages (Iglesias *et al.* 2014). Uncertainty over the nature and extent of change to fire regimes and in particular fire frequency and intensity creates challenges for managing ecosystems that have altered structure and function under climate change (Colloff *et al.* 2016). Due to this the management of fires has challenging issues that need to be faced in the future, particularly the role of climate change and the potential of increased frequency in wildfires (Keane and Loehman 2009).

Fire plays an important ecological role in maintaining biodiversity on Fraser Island and with projected climate change it is expected that there will be an increase in the duration and severity of the fire season resulting from the decrease in moisture and the increase in temperature (Wardell-Johnston *et al.* 2015). Result from Spencer and Baxter (2006) strongly suggest that frequent fires on the Island have resulted in a decrease in relative diversity through the dominance of several species such as bracken (*Pteridium esculentum*), grasses (*Themeda triandra* and *Imperata cylindrica*), restiads and heathland species (*Coleocarya gracilis*) and *Xanthorrhoea* spp.

6.2. Study Region

Fraser Island (K'gari the Island's Aboriginal name), lies along the eastern coast of Australia and is the world's largest sand island that developed over the last 800,000 years (Peace 2001; Yizhaq *et al.* 2013; Gontz *et al.* 2015; Moss *et al.* 2015; Stewart and Moss 2015) (Figure 6.1). The importance of Fraser Island has been highlighted by a number of Queensland, Australian, and international environmental protection agreements, including World Heritage and Convention on Wetlands of International Importance (Ramsar) listings due to its unique natural values (Behrendorff *et al.* 2016). The island is situated between the tropical and subtropical boundary off the east coast of Queensland and is approximately 124 km in length and 20 km wide. Average annual rainfall is between 1300 to 1700 mm/year with mean temperatures between 14 °C in winter and 29 °C in summer (Donders *et al.* 2006). The soils are composed of marine and aeolian sand deposits with few scattered bedrocks of basalt, with the central portion of the island being composed of complex sand dunes that are vegetated, with numerous lakes occurring in the depressions between the dunes (Gontz *et al.* 2015).

There are presently approximately 1246 recorded plants on Fraser Island in the Great Sandy Region National Park (DSITIA 2014) that comprise of five main community types, (i) rainforest, predominantly notophyll vine forest, including wet sclerophyll forest (Figure 2, b, d and h), (ii) tall open forest (Figure 6.2, e), (iii) low open forests and woodlands (Figure 6.2, a and e), (iv) heathlands from wet to dry heath (Figure 6.2, c), *Empodisma minus* wetlands (Figure 6.2, f) (v) and *Melaleuca* forest (Figure 6.2, g). Coastal communities include mangrove forest and salt marsh, which are not relevant to this study. The vegetation of Fraser Island is laterally zoned following the contours of successive parabolic dune systems that run parallel to the islands coastline (Donders *et al.* 2006), with younger parabolic dunes to the east. The vegetation follows the contours of the parabolic dune systems that are parallel to the coast with older dunes occurring in the west of Fraser Island. Nutrient availability increases with the age of the sand and amount of detritus initially then declines due to leaching through a retrogressive succession process. (Walker *et al.* 1981; Longmore and Heijnis 1999; Donders *et al.* 2006).

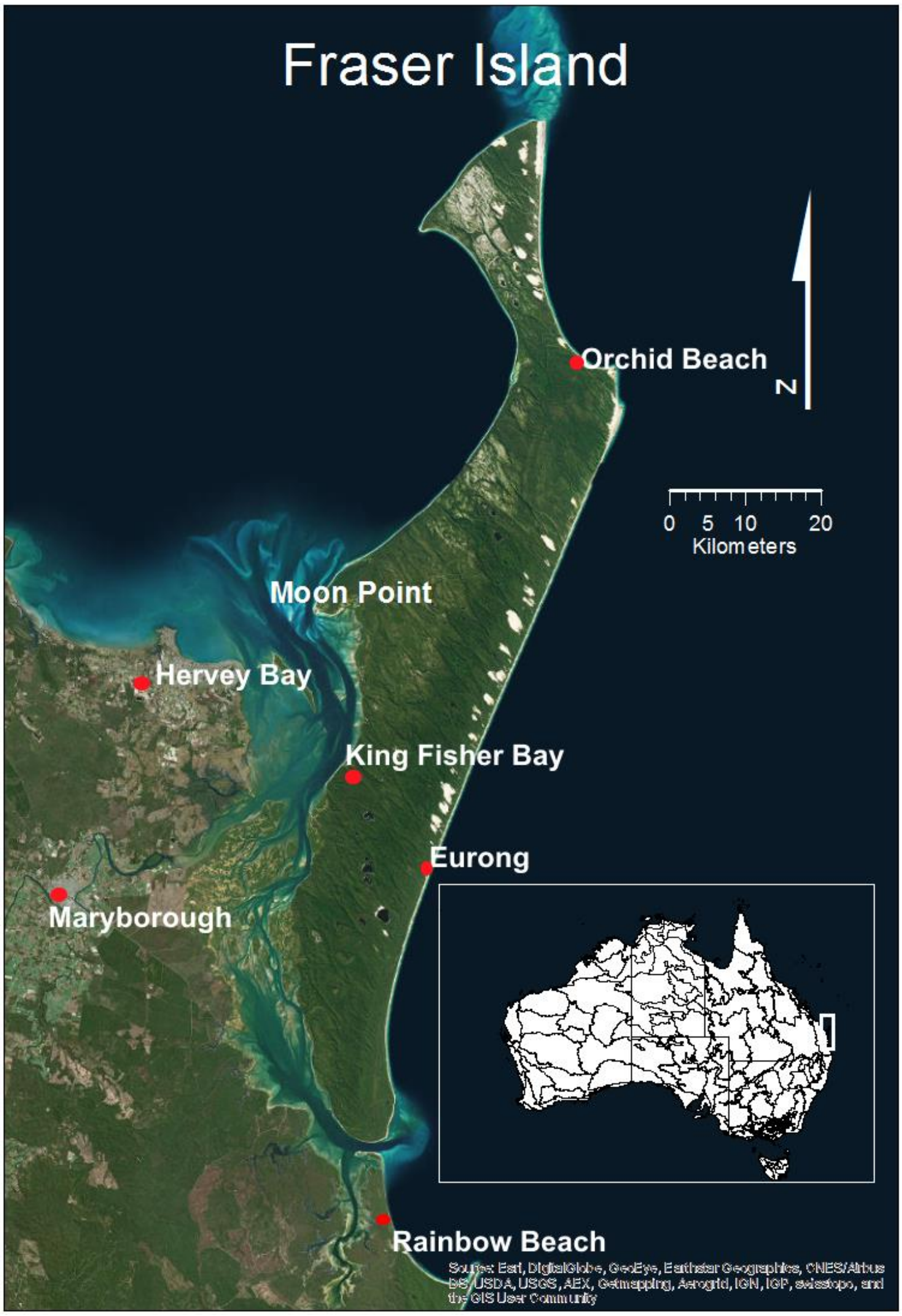


Figure 6.1: Map of the Great Sandy Region with showing the location of Fraser Island within this region.

According to Sinclair (1997) there are 6 dune types based on age development that play an important role in vegetation composition, where dune type 1 is bare sand and pioneer plants moving to more complex vegetation at dune type 4 of tall forest. Dune type 5 being much older and nutrient poor supporting smaller and less vegetation such a stunted woodlands and dune type 6 being the oldest dune type and most nutrient poor resulting in retrogressive succession influencing vegetation composition such as shrubs and low heathlands.

Figure 6.2: Vegetation community types showing (a) open forest (b) Brush Box forest (c) heathlands (d), Rainforest (notophyll vine forest) (e), open forest and woodlands (f), Wetlands and Empodisma fens (g), Melaleuca forest (h) and Syncarpia forest.

Photographs: Philip Stewart

6.3 Methodology

A fire simulation model FireBGCv2 landscape fire succession model was downloaded and installed on a high performance computer. FireBGCv2 is a research simulation platform for exploring fire and vegetation dynamics that has five inherent scales built into its design that correspond to spatial domains or organizational layers represented within the model (Figure 6.3). Fire management and climate change are important factors for modern landscape management, and as such, new means are needed to address these challenges. Field studies, while preferable and reliable, are problematic because of the large time and space scales involved. Therefore, landscape simulation modeling has more of a role in wildland fire management as field studies become less tenable. FireBGCv2 is a C++ computer program, which is intended as a research tool (Keane *et al.* 2011) and may be used as a predictor of change.

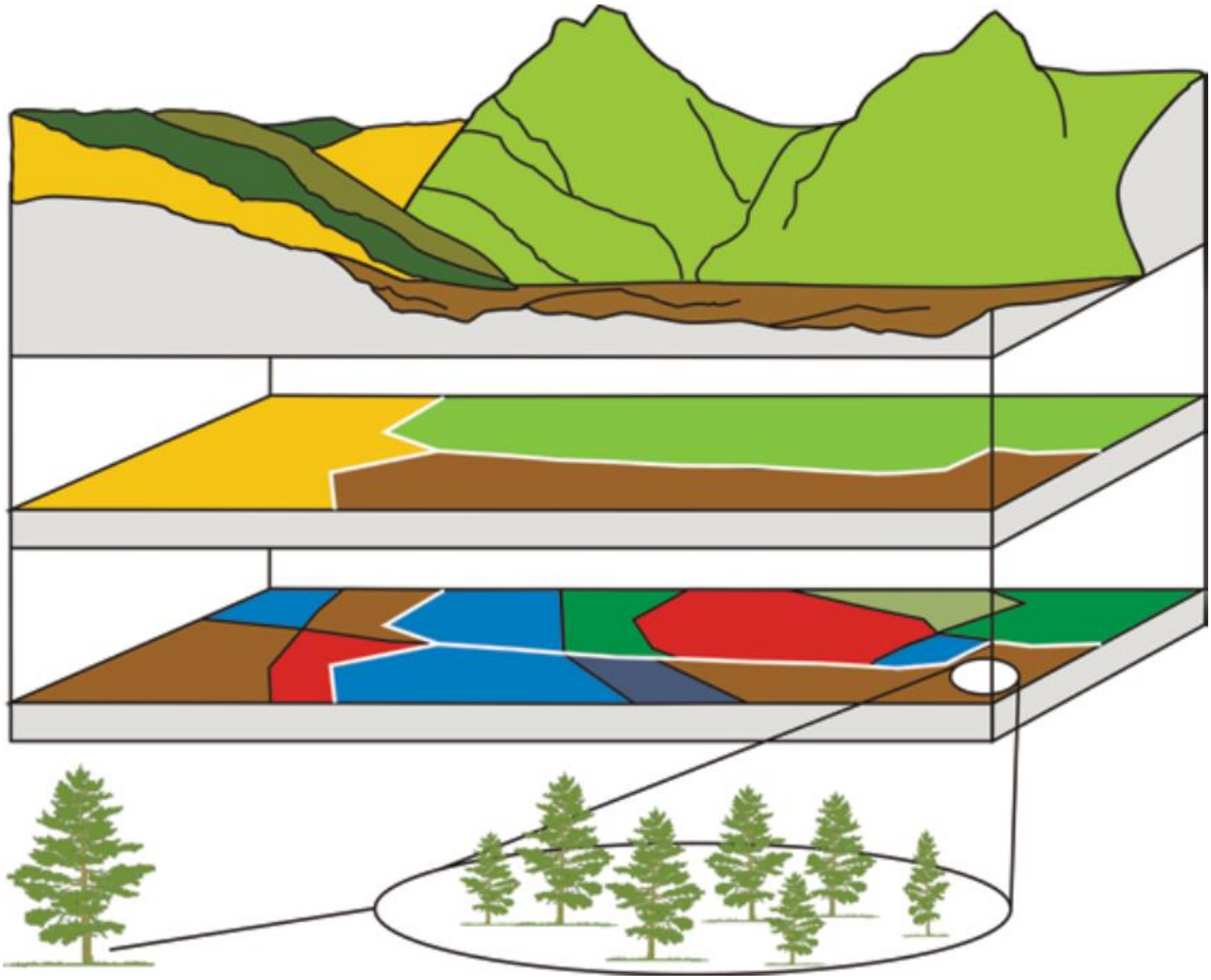


Figure 6.3: The five organisational scales built into the design of FireBGCv2: landscape, site, stand, species, and tree. Source: Keane *et al.* 2011

There are two main types of files in the FireBGCv2 and these are, (i) the parameter files, which are scenario based files and subjective (ii) and input files that are description, representing the site, stand and tree conditions. Input files describe the actual ecological conditions gathered from field data collection. Most input files are vertical and all output files are horizontal (Keane *et al.* 2011). To initialise and execute FireBGCv2 a Driver.in file, which is a discrete scenario file containing the names and pathnames of all input files, is required and the order of these files is critical. The Sim.in file contains general simulation parameters for the execution of the program and serves as error checks in some files.

A rigorous literature review was undertaken for parameters to run scenario based assessments. In addition to this, two field trips totalling 6 days were undertaken to collect and log descriptive vegetation data from the 25th to 29th of April 2015 and again from the 16th to 20th July 2015. Data collected was inputted onto a plot description form that was collected for each of the 12 randomly selected plots of each vegetation type across Fraser. The overall objective for the field sampling was to collect vegetation and fuels data for parameterisation of FirebGCv2 that needs to be input into the Species.in file.

White *et al.* (2000) was used for the physiological parameters that established the model for Fraser Island as a new region. To this end, 12 stands were sampled representing the major vegetation guilds (Table 6.3) that comprise the simulation area of Fraser Island. Each survey sight was sampled for vegetation (over-story and understory) and fuels within a nested circular plot (Figure 6.4). Representative vegetation types were defined as combinations of dominant over-story vegetation and successional stage (Figure 6.5). The general method for sampling (Table 6.1) the plots was to opportunistically locate a plot within a representative stand type within the study area, defined as combinations of dominant over-story vegetation and successional stage (full sampling protocol Appendix 1). For example, stand cover types within Great Sandy Region are classified as combinations of dominant tree species within successional stages that represent time since last disturbance.

Table 6.1: Data collected within nested, hierarchical plots as shown below.

Data type		***Plot radius/area***
Trees	Seedlings only	3.6 m/.004 ha
	Saplings through Mature	11.34 m/.04 ha
Understory (herbs, forbs, grasses)		11.34 m/.04 ha
Fuels (Photoload sampling)	1-1000 hour	4 x 1 m^2 plots at 5 m
	Herb and forb biomass	4 x 1 m^2 plots at 5 m
	Fuel depths	2 corners in each 1 m^2 plot

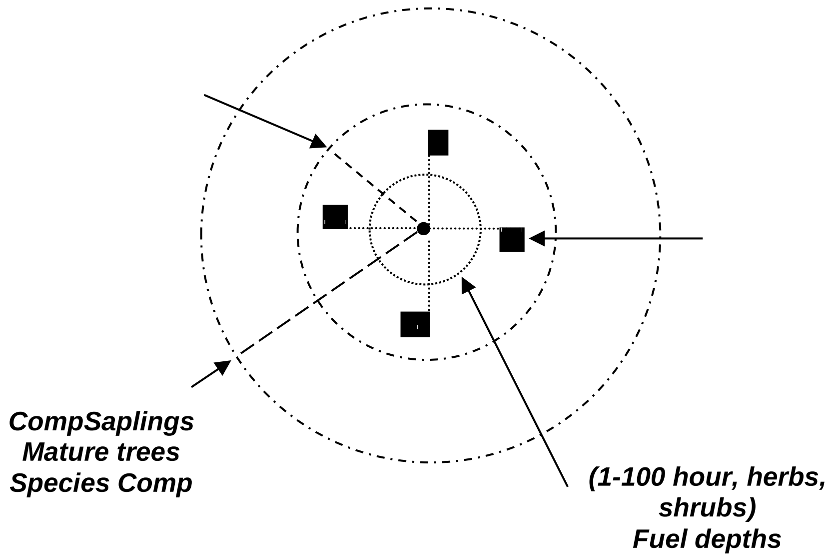


Figure 6.4: Diagram of sampling methodology for each plot as set-out in Table 6.1.

FireBGCv2 requires three map layers or ASCIIGRIP files for execution for the Map.in file. The first is a digital elevation model (DEM) with cell size 25 by 25 metres and height 242.76 metres, the second input map layer is the initial stand map and the third input map layer is the zone map. The DEM for Fraser Island was downloaded from Geoscience Australia and using ArcMap (ESRI software) the island was clipped out and converted to ASCIIGRID through the conversion tool, ArcMap. The stand map data that spatially represents all the simulation stands on the landscape for Fraser Island was obtained from the Queensland State Government and reclassified into 12 guilds (Table 6.2), then converted into raster data. The zone map data was created using the stand data and reclassified into three data types, height, aspect and vegetation type (forest/non-forest). The zone and stand maps are overlayed to create the stand polygon map, with each stand assigned a unique stand ID number. The stand polygon map is then converted to a raster data map. All three map layers share a common map area and pixel size of 25 metres^2, converted to ASCIIGRID.

The vegetation data obtained from the Queensland State Governments Regional Ecosystem Description Data (REDD) where there are 13 bioregions recognised in Queensland with South East Queensland (SEQ) as bioregion 12. REDD supersedes the regional ecosystem descriptions in Sattler and Williams (1999) and includes updated descriptions, to improve clarity, including additional regional ecosystems and vegetation communities recognised since 1999 (Queensland Herbarium 2015). The regional ecosystem framework is composed of three parts, a bioregion, land zone and vegetation, each with a unique code. The regional ecosystem for the vegetation of Fraser Island was downloaded and manipulated in ArcMap to reclassify and recode vegetation into the 12 guilds (Table 6.2) from a total of 2409 regional ecosystems utilizing Microsoft Access to link the data bases for use in ArcMap. Data were converted to ASCII format for FireBGCv2.

A total of 12 broad vegetation classifications or guilds (Table 6.3) were created for the stand map of the major vegetation types of Fraser Island by reclassifying the regional ecosystem data into very broad similar vegetation types (Figure 6.7). Guilds were identified as being either able to burn or not, which resulted in whether or not a guild would be surveyed (Table 6.3). Each guild that would burn was surveyed according to the sampling protocol. A guild that was less 2.0 km^2 or 100 hectares in area was deemed not to be a valid plot within a stand due to the requirements of FireBGCv2 plot size.

A daily weather data file is specified for each of the 12 sites that were included in FireBGCv2 Simulations that contains daily weather data for all years taken from a weather station at or near each site. The method used was to run the MT-CLIM, which is a computer program that uses observations of daily maximum temperature, minimum temperature, and precipitation from one location (the "base") to estimate the temperature, precipitation, radiation, and humidity at another location (the "site"). MTCLIM generates weather information for another location with potentially different elevation, slope and aspect from the input location.

The input location is referred to as the "base station", while the new location for output is referred to as the "site". Required input data from the base station are daily observations of maximum and minimum temperature, and daily total precipitation (NTSG 2016). Base station weather data from January 1969 to May 2015 were obtained from the Bureau of Meteorology's Sandy Cape Lighthouse (039085) situated on the northern end of Fraser Island, latitude: 24.73 degrees South and longitude: 153.21 degrees East, elevation 99 metres.

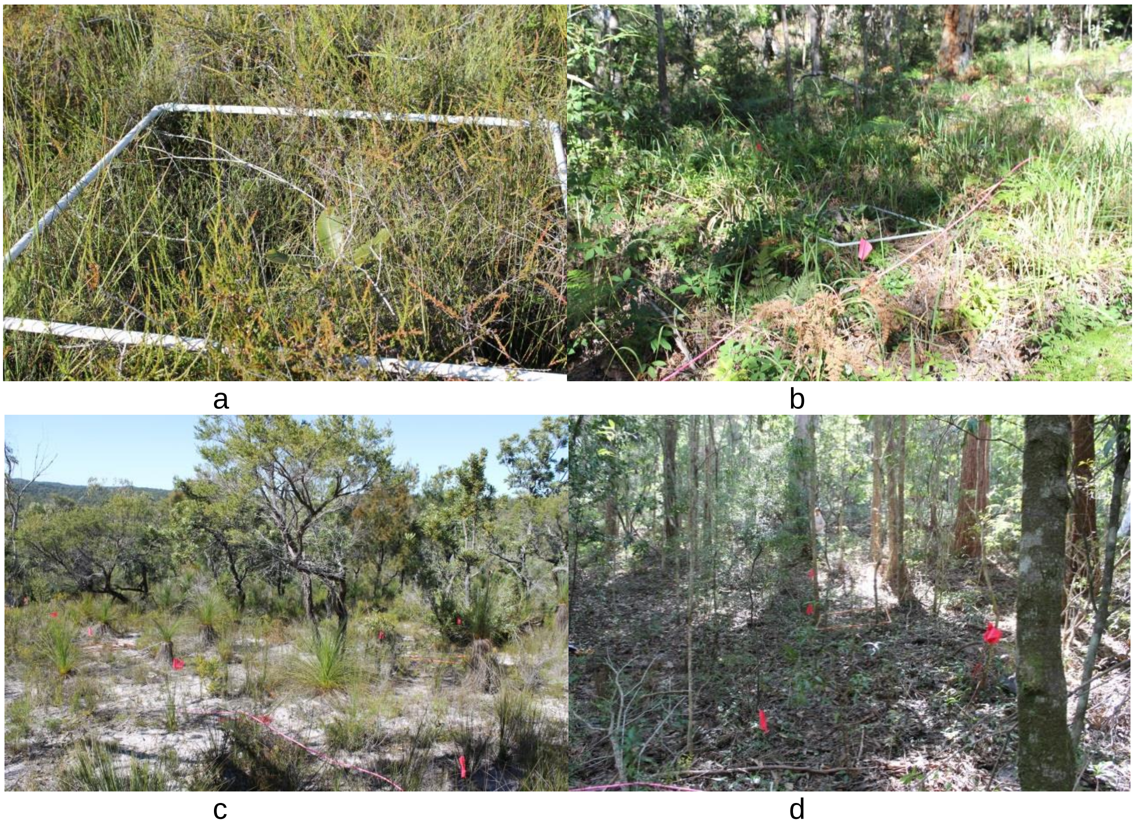


Figure 6.5: Examples of the representative vegetation types surveyed. Empodisma sedgelands sample plot showing the density of vegetation within the 1^2 m transect (a), demonstrating the method of using rope (pink line) to measure distance within sample plot in open forest woodlands

(b), heathlands flagged for sampling showing heath composition (c), Syncarpia and Brush Box forest surveyed using ref flags and rope (d).

6.4 Results

Test runs of FireBGCv2 were undertaken using hypothetical data to ascertain if it was possible to initialise and execute the fire simulation model on the available computer systems. Initial errors occurred due to inputting incorrect file paths, however once these issues were resolved FireBGCv2 ran without any fatal errors (Figure 6.6).

Having completed a number of test initialisations and executions of FireBGCv2, the focus turned to developing the maps required for model runs. A substantial amount of time (13 months) was spent on developing the DEM, as well as the stand, site and zone map data. The DEM and zone data were developed without any issue and converted into ASCIIGRID and placed in the Map.in file. Problems occurred while creating the stand.asc map as it requires a site map that divides Fraser Island into similar sites based on soil, weather and topography. The site map is then used with the stand map by overlaying the stand map over the site map to create the stand polygon map thus creating a site-stand combination that is described in the stand.in file.

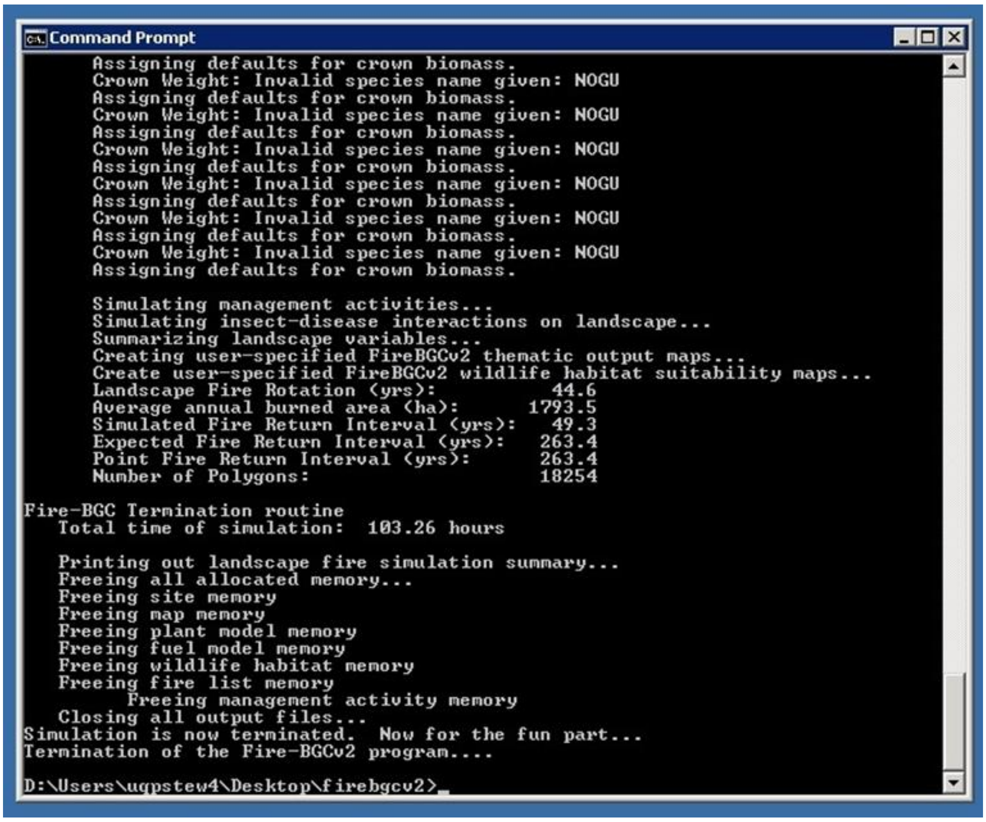


Figure 6.6: A test run with any errors of FireBGCv2 using hypothetical data with a total simulation time of 103.26 hours.

Vegetation physiological parameters collected during the field trips in April and July 2016 were inputted into a parameterisation excel file (Table 6.4) for recording physiological parameters, which is then inputted into the species.in, plant in, Fuel.in (Table 6.5) and other applicable files.

Due to the complexity of the vegetation types, topography, and extent of Fraser Island landmass it was found that more time would be required to parameterise FireBGCv2. Creating mapping data and the lack of sufficient information and publications on initialising and executing the model it was not possible to initialise and execute the model on any Fraser Island data. Although extensive research and time was spent on creating map data (Figure 6.7), it was not possible to run the model, even though other initialisation and execution requirements were met.

Table 6.2 Vegetation guilds developed for use in FireBGCv2 landscape fire succession model of Fraser Island vegetation. Guilds of less than 200ha are highlighted as the minimum size for a plot is $2km^2$. Guilds that are to burn are listed Y and guilds not to burn are listed N

Guild	Ha	Survey	burn	veg_type	RE1	biodivstat
1	19621.9	No	N	Wetland	12.1.2	No concern
2	11539.52	No	N	Sand	12.2.14	No concern
3	1068.72	Yes	Y	Open heath	12.12.19	Of concern
4	4031.36	No	N	Notophyll vine forest	12.2.1	Of concern
5	2300.36	Yes	Y	Open forest leached soils	12.2.11	No concern
6	68595.61	Yes	Y	Empodisma sedgelands	12.2.15	No concern
7	47745.24	No	N	Araucarian vine forest	12.2.3	Of concern
8	24.08	No	Y	Syncarpia forest	12.2.4	Of concern
9	9256.64	Yes	Y	Open forest dunes sand	12.2.6	No concern
10	926.73	Yes	Y	Coastal sandy plain	12.2.7	Of concern
11	155.82	No	Y	Plant	Plant	
12	1.35	No	Y	Non remn	Non rem	
	1 square kilometre = 100 hectares		**A valid plot location is located within a stand that is at least 2.0 km^2 in area**			

Table 6.3 Part of the finalised species input data file for fireBGCv2 showing species and plant physiological parameters information as an example.

Species 4-Letter Name	Common Name	Species ID Number	Vegetation Lifeform	Shade Tolerance Class	Sprouting Ability	Rain Interception Coefficient	Light Extinction Coefficient	All-sided-to-projected Conversion Factor	SapwoodC to DeadwoodC Ratio	SLA	Shade:Sunlit SLA	Fraction of leaf N in Rubisco	gmax	gc	gb	Leaf Water Potential at field capacity	Leaf Water Potential at Stom	Min Vapor Pr
EUSB	Scribbly Gum	01	0	2	3	0.063	0.5	2.0	0.9	32	2	0.04	0.0060	0.00006	.01	-1.0	-3.0	1000
EUPS	Blackbutt	02	0	2	3	0.063	0.5	2.0	0.9	32	2	0.04	0.0060	0.00006	.01	-1.0	-3.0	1000
MOSC	Prickly broom heath	03	0	4	1	0.045	0.51	2.8	0.9	20.0	2	0.04	0.0060	0.00001	0.08	-0.50	-1.30	500
CACO	Cypress pine	04	0	4	1	0.045	0.51	2.8	0.9	20.0	2	0.04	0.0060	0.00001	0.09	-0.50	-1.30	500
MEQU	Paperbark tea tree	05	0	4	1	0.0375	0.6	2.0	0.9	32	2	0.04	0.0060	0.00006	.01	-0.34	-2.2	1200
ELRE	Blueberry Ash	06	1	5	1	0.0375	0.61	2.0	0.9	32	2	0.04	0.0060	0.00006	.01	-0.34	-2.2	1200
ACLE	Black Wattle	07	0	4	0	0.0375	0.54	2.0	0.9	32	2	0.04	0.0060	0.00006	.08	-0.34	-2.2	1200
BASE	Red Honeysuckly Banksia	08	2	2	2	0.045	0.55	2.0	0.9	12	2	0.04	0.0060	0.00006	.02	-0.81	-4.2	970
LEPO	Common Teatree	09	2	4	1	0.045	0.55	2.0	0.9	12	2	0.04	0.0060	0.00006	.02	-0.81	-4.2	970
SHRB	Heath	10	2	3	2	0.045	0.55	2.0	0.9	12	2	0.04	0.0060	0.00006	.02	-0.81	-4.2	970
CYSP	Sedge grass	11	3	1	3	0.0225	0.4800	2.0	0.9	49	2	0.04	0.0060	0.00006	.04	-0.73	-2.7	1000
POAS	Poa species	12	3	1	3	0.0225	0.4800	2.0	0.9	49	2	0.04	0.0060	0.00006	.04	-0.2	-4.0	1000
RUSH	Rush species	13	3	1	3	0.0225	0.4800	2.0	0.9	49	2	0.04	0.0060	0.00006	.04	-0.2	-4.0	1000

Table 6.4: continued. Species four letter code, common and scientific names including species ID number for FireBGCv2.

Species 4-Letter Name	Common Name	Scientific Name	Species ID Number	
EUSI	Scribbly Gum	Eucalyptus signata	01	
EUPI	Blackbutt	Eucalyptus pilularis	02	
MOSC	Prickly broom heath	Monotoca scoparia	03	
CACO	Cypress pine	Callitris columellaris	04	
MEQU	Paperbark tea tree	Melaleuca quinquenervia	05	
ELRE	Blueberry Ash	Elaeocarpus reticulatus	06	
ACLE	Black Wattle	Acacia leiocalyx	07	
BASE	Red Honeysuckle Banksia	Banksia serrata	08	
LEPO	Common Teatree	Leptospermum polygalifolium	09	
SHRB	Heath	Shrubland	10	
CYSP	Sedge grass	Cyperus sp	11	
POAS	Poa species	grasses	12	
RUSH	Spreading rope rush	Empodisma minus	13	

6.5 Discussion

There has been a surge in the development of fire models over the past ten years as part of a wider effort to model the processes driving fire dynamics in real landscapes where fire models reproduce fire regime attributes for assessment of fire impacts on different scales (Duane *et al.* 2016). Models that predict future fire regimes depend on relationships between annual temperature and fire activity that are based on historical studies correlating annual climate variation with fire activity, and at different spatial scales (Keeley and Syphard 2016). However according to Keane *et al.* (2004) a major difficulty in predicting large-scale ecological change is the inclusion of non-equilibrium dynamics, disturbance regimes and spatial relationships into fire simulation models.

FireBGCv2 landscape fire successional model is complex mechanistic model that can take years to run a simulation using complex simulation designs (Keane *et al.* 2011). The complexity of the model and especially mapping data is a major factor and requirement for model initialisation and execution. It is organized around five hierarchical representations of vegetation and landscapes and simulates process at each of these levels such as from finest to coarsest, which are the levels of the individual tree, species, stand, site, and landscape (Yospin *et al.* 2015). The physiological parameters of the vegetation of Fraser Island were drawn from a number of sources, however precise parameters were not available for most of species (Table 6.4) and as such had to be inferred from closely related species where empirical observations are available in White *et al.* (2000).

FireBGCv2 does not have a manual and there is no formal support provided for the model. It is provided free on the condition that these terms are accepted. There are limited publications on the model and examples of simulations. One such publication and probably the most important (i.e. closest to being a manual) is the paper by Keane *et al.* (2000) *The FireBGCv2 Landscape Fire Succession Model: A Research Simulation Platform for Exploring Fire and Vegetation Dynamics*. Information on parameterisation, initialisation, calibration, simulation and output are provided. However creating mapping data and the development of such is only discussed in brief. Little if any information on how to create or develop this data is available and what there is can be confusing (Keane *et al.* 2011).

The creation of mapping data is vital for FireBGCv2 initialisation such as defining a set of sites across the study area where site boundaries are based on similarity of soil, geology and climate for example. Stand maps must define the dominant vegetation types within sites (Figure 6.7). Overlaying of these two map data types provides the stand.asc data for use in the model. However the process or methodology in creating such is not provided and limited information available on how to proceed with this map creation. Unfortunately by not being able to initialise and execute FireBGCv2 within the time limits of the research project, it was not possible to access any fire scenarios or projections of future climate change on fire on the landscape. However the Fraser Island landscape is a shifting mosaics of successional communities created or maintained by disturbance regimes through complex interactions and feedbacks between fire, vegetation, fuel, and climate. This provides a unique opportunity to examine these relationships and how they may respond to changing climates (Keane *et al.* 2013).

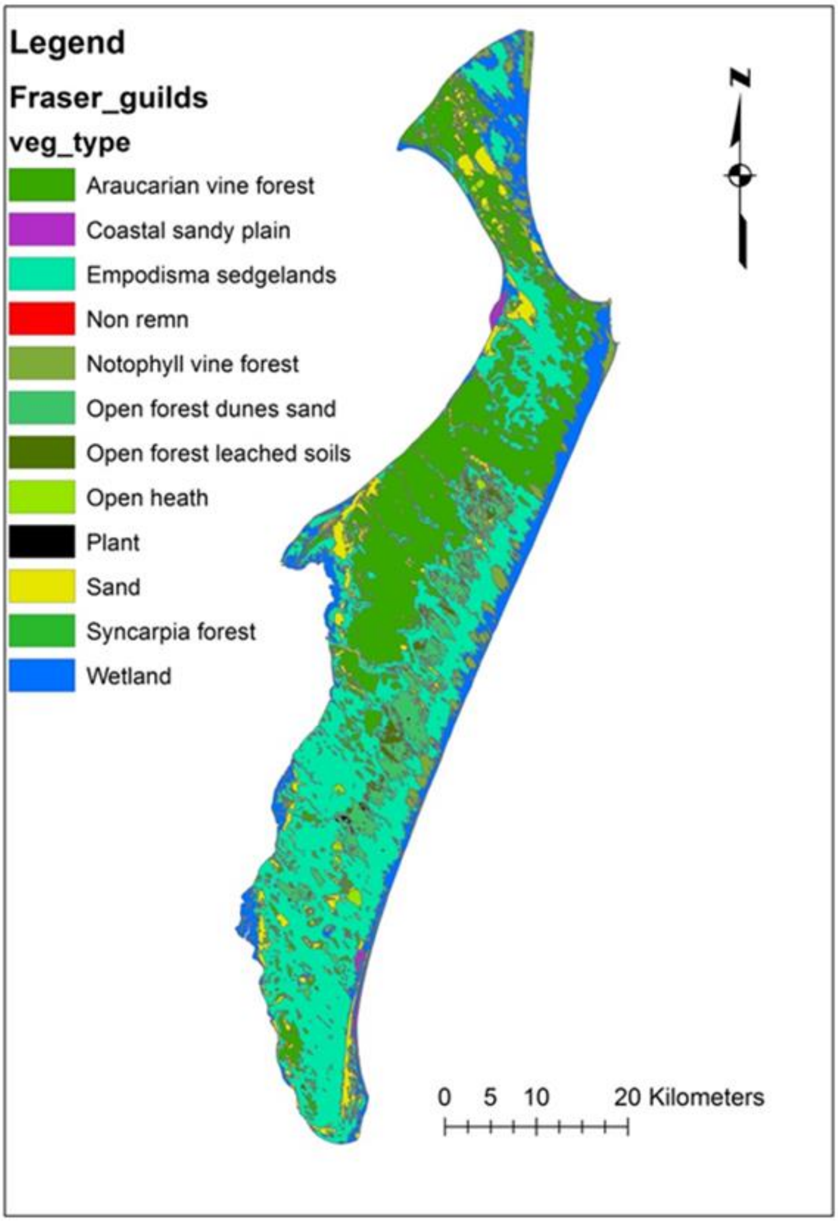
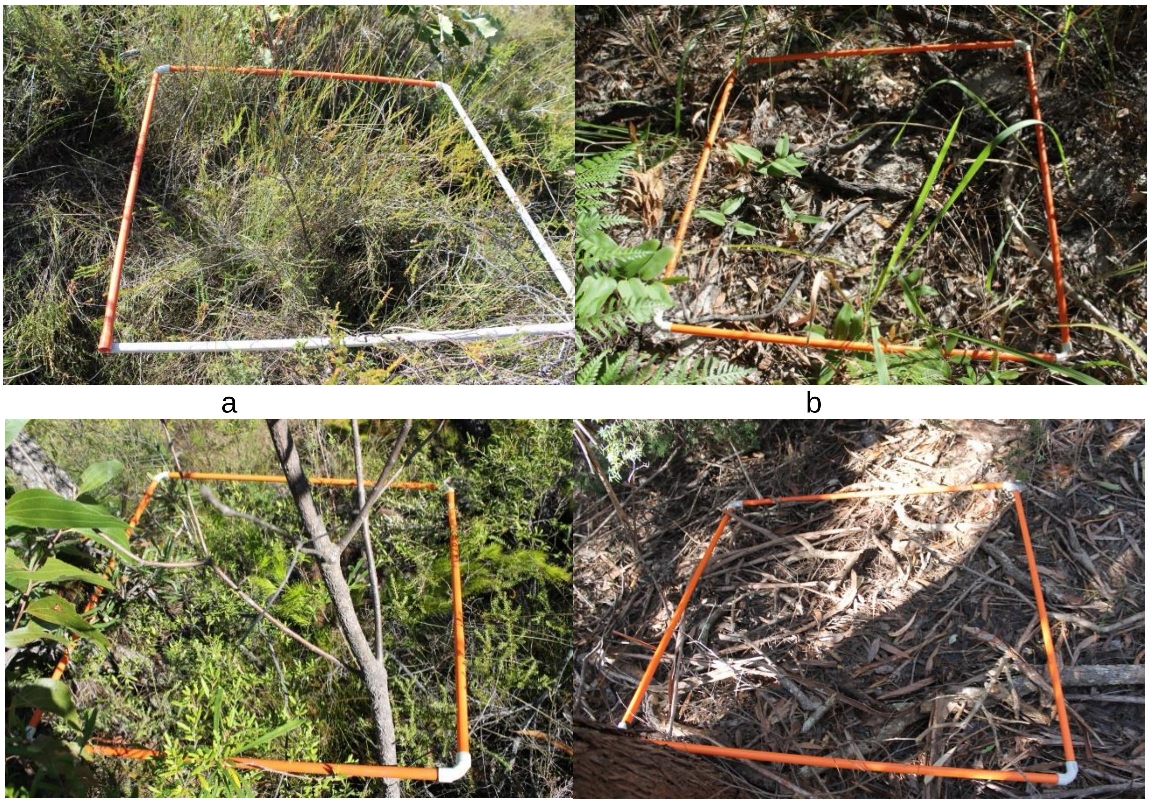


Figure 6.7: Map developed of Fraser Island's 12 vegetation guilds created for FireBGCv2 fire simulation model.

Field sampling provided data for vegetation and fuel parameterisation of FireGBCv2 where fuel loads were assessed and recorded according to their characteristics (1 hour, 10 hour, 100 hr and 1000 hr fuels). 1 – 100 hour fuels are recorded by photoload in sequence (a photograph is taken of each 1 x 1m^2 plot where fuel loading is then estimated. All logs > 7.6cm have the small and large diameters are measured and recorded as 1000 hour fuels (Appendix 1). Each vegetation stand having a site specific fuel characteristic. For example *E. minus* wetlands produced mostly 1 to 10 hrs fuels, with 100 hr fuels on the boundary, open heath with 1 to100 hr fuels and 1000 hr fuels on the boundary with forest or woodlands. Open forest fuels were a mix of 1 to 100 hr and 1000 hr fuels. From this it is deducted that heaths and *E. minus* wetlands would burn faster and be of a shorter duration than forests and woodlands, further fire intensity would vary in heath due to age, the patchiness of the vegetation and amount of bare ground in many stands where there is limited fuel, compared to open forest or woodlands that tend to have larger fuel types subjecting the stand to longer duration burns (Figure 6.8).



a b c d

Figure 6.8: Photoloads of 1 to 1000 hour fuels. (a) Empodisma wetlands showing high 1 to 10 hr fuel load and density of vegetation, (b) Scribbly gum open forest fuel loads showing a mix of 1 to 100 hr fuels with scattered 1000 hr fuels outside of the quadrate. (c) Open heath with very high 1 to 100 hr fuel loads, there is very limited 1000 hr fuels scattered on the stand boundary. (d) Eucalyptus and Monotoca forest showing high 1 to 1000 hr fuels.

The duration of burn and fire intensity (Hosseini *et al.* 2017) will affect vegetation and soils and is dependent on fuel size, load, distribution and density (Pellegrini *et al.* 2017). From field samples loads in respect of 1 to 100 or 1000 hr fuels varied little within the same stand types with similar types, however density varied greatly between all stands, which depended on the stand vegetation structure and type. However with expected shifts in vegetation from *E. minus* wetlands to sclerophyllous forest for example (Chapter 5) it is expected that heavy fuel loads will increase, potentially increasing fire intensity and changing fire frequency. These changes will be compounded by a changing climate, which can only be further compounded by invasive vegetation such as grasses (Chapter 3), which in turn will increase fuel loads and fire frequency transforming the ecosystem once again.

The author will continue to work on FireBGCv2 as it is felt the model has substantial value to management for projecting possible fire scenarios, which will only become more important and valuable as a management tool as climate change intensifies.

Table 6.5: Fuel model parameters for fire behaviour and fire effects simulation. Each line has eight values for each parameter for each fuel component (Duff, litter etc) Keane *et al.* 2011.

Fire-BGCv2 -- Fuel model parameters (P. Stewart, GPEM, UQ, 06/2016)								
Fuel Comp	Duff	Litter	1 hour	10 hour	100 hr	1000 hr	Shrub	Herb
		1 Fuel Model Number - Low elevation FBM 8						
rhop-dead	0.6	0.510	0.390	0.390	0.390	0.390	0.510	0.510
lhv-dead	18586.7	18586.7	18586.7	18586.7	18586.7	18586.7	18586.7	18586.7
mps-dead	111.00	57.410	61.166	11.760	2.880	0.980	3.156	91.856
moistdead	0.60	0.060	0.060	0.080	0.100	0.100	0.100	0.050
consume-dead	0.90	0.900	0.950	0.845	0.845	0.790	0.900	0.990
rhop-live	0.55	0.510	0.390	0.390	0.390	0.390	0.513	0.513
lhv-live	18586.7	18586.7	18586.7	18586.7	18586.7	18586.7	18595.0	18595.0
mps-live	111.00	57.410	61.166	11.760	2.880	0.980	49.200	91.860
moistlive	0.60	0.800	0.060	0.080	0.100	0.150	0.800	0.500
consume-live	0.90	0.900	0.900	0.800	0.500	0.010	0.900	0.900
mext	0.30	0.300						
bulk-d/l	0.01	0.001						
duflitbulk	76.90	44.10						
fdepth	0.16							
spreadcomp	1.0							
windmultfactor	1.0							

6.5.1 Limitations of Models

Simulation models can be categorised into four approaches, i) empirical models that build on statistical relations from actual data, ii) deterministic models that use generalised functions to look at relationships, iii) stochastic models that use probability distributions to represent ecosystem functions and iv) mechanistic models that use biological and physical relationships to simulate system behaviour, with each approach having advantages and disadvantages as shown in Table 6.6 (Keane *et al*. 2011). Managers need tools such as models to predict how changes will affect landscape composition and ecosystem function; however a robust mechanistic approach is not possible at landscape scales according to Gustafson (2013).

Table 6.6. Advantages and disadvantages of the approaches used in simulation models. (Keane *et al*. 2011).

ATTRIBUTE	EMPIRICAL	DETERMINISTIC	STOCHASTIC	MECHANISTIC
Complexity	Low	Low	Moderate	High
Parameter Requirements	Low	Moderate	Moderate	High
Accuracy	High	Variable	Low	Variable
Exploratory uses	Low	Moderate	Moderate	High
Management application	High	High	Low	Moderate
Portability to other situations	Low	Moderate	Moderate	High
Expand -ability	Low	Moderate	Moderate	High
Required Expertise	Low	Moderate	Moderate	High
Computer requirements	Low	Moderate	Moderate	High
Preparation time	Low	Low	Moderate	High

Models need to simplify real world processes such as the FireBGCv2 model that simplifies or excludes some biophysical processes affecting disturbances (Loehman *et al*. 2016). FireBGCv2 fire simulation model was developed to simulate fire and succession on Rocky Mountain landscapes and to operate at the high end of complexity and concreteness (McKenzie and Perera 2015). Further the model can be difficult to parametrise, initialise and execute (Keane *et al*. 2001). McKenzie and Perera (2015) state that model behaviour can become unstable nonlinear and inexact (Keane *et* al. 2011) due to running of hundreds of parameters with a further limitation of cumulative uncertainty from multiple approximations in complex processes.

One important limitation of FireBGCv2 fire simulation model, which is a mechanistic model is the lack of a linkage between fire spread and fire behaviour with fuels and weather (Keane *et al*. 2011). Fire behaviour by nature is none-linear making it difficult to make a valid quantitative statement regarding the relationship between input data accuracy and output accuracy (Alexander and Cruz 2013). Further input of fuels in models use specific formats requiring details such as locations and dimensions of individual trees, spatial distribution of understory fuels surface area and moisture content, making data collection difficult and time consuming (Pimont *et al*. 2016).

Algorithms, parameters, and assumptions in FireBGCv2 represent our limited understanding of the complex interactions among biotic and abiotic ecosystem components (Clarke *et al*. 2017). According to Keane *et al.* (2011) results from model runs are influenced by stochastic elements of fire spread, tree regeneration and parameter settings including simulation design. Further Mann *et al* (2016) showed that wildfires respond to changes in vegetation through the introduction of invasive grasses for example however this is not captured in their empirical wildfire model. A review of the literature shows that there is limited information on the limitations or disadvantages of mechanistic spatially explicit models and especially fire simulation models such as FireBGCv2.

6.6 Conclusion

Fire management faces many challenges with climate change and changing fire regimes, with modelling being an important tool for fire research and management, offering an effective method to explore and evaluate management actions and ecological change (Keane *et al.* 2011: Keane *et al.* 2016; Loehman *et al.* 2016). Models can be used to simulate effects of alternative fire treatments to determine effective fuel reduction methods or ecological restoration strategies (Keane *et al.* 2011). Although it was not possible to initialise and execute FireBGCv2 outside of test runs due to complexities of the spatial data and extensive set of ecophysiological parameters that were not available for the selected ecosystem, it is felt that as a mechanistic, individual tree succession model containing stochastic properties implemented in a spatial domain and because it is the only model to have a close link between climate and vegetation growth (Keane *et al.* 2013) it has tremendous management value and further investigation into initialising and executing the model is recommended for the Great Sandy Region and in particular Fraser Island due to its complex vegetation, climate, soil relationships.

Chapter 7. Conclusion

7.1 Summary

The aims of this thesis was to; i) Identify impacts of invasive grasses, climate change and ecosystem transformation through increase fuel loads fire frequency and intensity, ii) produce a palaeoenvironmental reconstruction of fire climate and vegetation from the LGM to present identifying shifts and changes to past vegetation community structure, composition and distribution, iii) provide land change analysis data on vegetation thickening and encroachment due to reduced fire frequency and a change fire regime, iv) model fire scenarios for management in the present and possible future fire scenarios. The invasion and increased fuel loads, fire frequency and intensity providing insight into changes in vegetation dynamics and impacts of transformed ecosystems with implications for future management and possible impacts of climate change. Pollen and charcoal data provided insight into past fire events, climate and shifts in vegetation from rainforest to more fire prone sclerophyllous vegetation taxa, providing a foundation for contemporary vegetation. Land change analysis and quantitative vegetation analysis to identify vegetation thickening and encroachment through possible reduced fire frequency, and changed fire management. Lastly scenario based projections of fire, climate and vegetation dynamics through multiple temporal and spatial drives of change using a fire landscape simulation model FireBGCv2.

The research objectives of this thesis included:

1. Reviewing fire regime and fire histories of South East Queensland to identify past fire regimes of the region and associated plant communities.
2. Identifying changes to fire regimes and natural ecosystems through the invasion of exotic grass species and increased fuel loads.
3. Reconstructing past fire histories and fire regimes with associated vegetation communities shifts in the landscape from the Last Glacial Maximum to the late Holocene
4. Providing data on present day fire regimes with the associated plant communities and climate for comparison with the Paleofire and Paleoclimate data to identify any shifts or changes in wildland fire regime, climate and plant community structure.

5. Identifying and providing methods to project the past and present fire regimes and climate to identify possible future fire regimes, climate and plant community structure and any shifts in plant communities using a fire.

Objectives 1 is addressed in Chapter 2, which provided a comprehensive review of fire history and contemporary fire regimes and vegetation. Objective 2 is addressed in Chapter 3 where invasive grasses and climate change fire regimes by changing fire frequency and intensity with increased fine fuel loads creating transformed ecosystems. Objective 3 is discussed in Chapter 4 by reconstructing the fire history and vegetation shifts using charcoal and pollen records from the LGM, and Holocene. Objective 4 is addresses in Chapter 5 where land change analysis is undertaken to identify thickening of vegetation including forest encroachment through the reduction of fire frequency and changed management practices. Objective 5 is addressed in Chapter 6 in modelling of future fires and changes in vegetation dynamics and scenario based projections.

7.2 Key findings

This thesis identifies changes to fire regimes, changes to constituent vegetation species over time and space with impacts of a changing climate, invasion by exotic grasses, vegetation thickening and modelling of future scenario based fire management using a fire simulation model. Research presented in Chapter 3 on the changing climate, increased fuel loads and fire frequency resulting in ecosystem transformation with shifts in constituent species from savanna woodlands to grassland for example. Increases in fire frequency and intensity on savanna ecosystems is due to an increase in fine fuel loads made available by invasion of perennial grasses such as Guinea grass (*Megathyrsus maximus* syn. *Panicum maximum*), Gamba grass (*Andropogon gayanus*) and Mission grass (*Cenchrus polystachios* syn. *Pennisetum polystachion*). These grasses have caused dramatic changes to the fire regimes through a positive feedback cycle, called a “grass-fire cycle” (Setterfield *et al.* 2010; van Klinken *et al*. 2013; Wagner and Fraterrigo 2015) due to the increases in the abundance of fine fuels. With increases in fire frequency and intensity (D'Antonio and Vitousek 1992), a decrease in native tree and shrub cover and abundance can result, which will further facilitate more grass invasion, further increasing the risk of higher intensity and more frequent fires resulting in an ever increasing self-perpetuating fire cycle (Flory *et al.* 2015). Introduced grass invasion is likely to result in substantial changes to fire regimes in invaded ecosystems, as these fire promoting grasses are ecosystem transformers that have the potential to alter the community structure (Rossiter-Rachor *et al.* 2016).

Research presented in chapter 4 analyses macro charcoal and pollen that produces evidence of past fire events, and vegetation shifts with changes in climate and sea-levels including anthropogenic fire. Vegetation has shifted from mainly rainforest taxa to sclerophyllous fire adapted vegetation from 24 ka to 18 ka. Variations in climate have seen shifting vegetation patterns as the climate changes from cool moist to a warmer dryer climate. Sea level rise and permanent residence on Fraser Island combined with a dryer and warmer climate saw an increase in fire events with anthropogenic fires more than likely playing a pivotal role in the changing dynamics of fire on the Island. However with the arrival of Europeans there is a sharp spike in charcoal deposits and a rapid decline corresponding with the European fire exclusion policies.

Ecosystem transformation can also occur with a reduction in fire frequency and intensity, where fire is or has been withheld and there is a change in fire management, as Chapter 5 proves. Vegetation thickening and encroachment along the ecotone of an *E. minus* wetlands and sclerophyll eucalypt forest has resulted from a decrease in fire frequency, which has allowed woody taxa to take advantage of the reduced incident of fire to become established in an otherwise quasi-climax state within the *E. minus* wetlands. Changes in fire regimes have been shown to rapidly promote the increase of woody species in grasslands near forests (Swaine *et al.* 1992), and according to Moss *et al.* (2016) vegetation thickening is possibly a longer-term threat to the *E. minus* wetlands as the pollen record show an increase in arboreal taxa in post European sections of the record. As such this supports the hypothesis that a major cause of vegetation thickening is the result of moving away from traditional fire management practices in the region (Russell-Smith *et al.* 2003; Moss *et al.* 2016).

Invasion of natural systems by invasive grasses, ecosystem transformation and vegetation thickening are in part either drivers of or results of changing fire regimes and especially fire frequency and intensity. It is well documented that anthropogenic climate change is a major driver of fire (Harvey 2016; Liu and Wimberly 2016; Shirazi *et al.* 2017) and that fire intensity and frequency is correlated to periods of dry or moist weather events from the

LGM. Plants vary greatly in their response to fire as a result of their development of evolutionary traits that has led to fire tolerance and fire dependence and is a trait that evolved directly in response to fire (Hill and Jordan 2017). With future climate change and increase in fine fuels as a result of invasion by invasive species managers will need to rely on fire modelling to better understand future fire behaviour and to be able to evaluate alternative risk management strategies (Riley and Thompson 2017). Fire management faces many challenges with climate change and changing fire regimes, with modelling being an important tool for fire research and management, as it offers an objective and effective method to explore and evaluate management actions and ecological change (Keane *et al.* 2011: Keane *et al.* 2016; Loehman *et al.* 2016). Chapter 6 examines fire and fire modelling to provide managers of Fraser Island and the Great Sandy Region with a tool to assess the possible impacts and changes to ecosystems by wildfire, prescribed burning and climate change. According to Wardell-Johnson *et al.* (2015) the interactive effects of climate change and other threatening processes such as wildfire will have greater impacts on Fraser Island than climate change alone.

7.3 Limitations and recommendations for future research

This thesis identifies the relationship between fire, climate, vegetation and anthropogenic fire. Paleofire and vegetation histories identified through sediment core chronologies provide for reliable although not usually precise dating of fire events and shifts in vegetation. However interpretation of charcoal accumulation to recreate fire histories and past fire events can be frustrated by hiatus within the core chronologies for macro charcoal analysis. Further research on charcoal deposits of a wider area incorporating all of the island's *E. minus* wetlands could possibly provide interesting results of both local and regional fire events and histories for the region. Further, the relationship of fire with climate and vegetation could be looked at in increased resolution both spatially and temporally by looking at other *E. minus* records from other areas across subtropical and tropical Australia, as this taxon occurs from Byfield National Park in central Queensland to coastal locations in northern New South Wales.

Fire frequency and intensity are major drivers for ecosystem transformation especially with an increase in fine fuels. With climate change, invasion by invasive grasses, increased fuel loads and changes to fire regimes, ecosystems degrade and transform with a loss in species richness and biodiversity. A key threatening process to biodiversity within the tropics and subtropics is the invasion of natural systems by imported pastoral grasses that are invasive and changed the dynamics of natural fuel loads and fuel distribution increasing wildfire intensity and frequency. High frequency and intense fires caused by increased fuels change the dynamics of the ecosystem and landscape creating a grass-fire cycle, where native species are unable to perpetuate or reproduce as these fires exceed the of fire survival strategies that native species evolved with. Limitations are in modelling of future fuel load and distribution scenario and climate change of other introduced pastoral grasses that pose an invasive risk factor. Further research into threats posed by perennial introduced grasses in relation to fuel loads and distribution and future fire intensities is required to better understand implications of transformed landscapes.

Vegetation thickening and encroachment due to changes in fire frequency, intensity and management has been identified and may result in ecosystem transformation. As woody taxa encroach and expand into *E. minus* wetlands there is an accompanying shift in ecotone and species dynamics. However it should be pointed out that invasion of the *E. minus* wetlands by forest and *Banksia* will be limited in that waterlogged areas will not become sclerophyll forest. Using land change analysis software such as TerrSet (Idrisi) and undertaking manual vegetation quantitative analysis, thickening and encroachment have both been identified within the ecotone of Moon Point *E. minus* wetlands and peripheral sclerophyll forest over a 52 year period. Further analysis extended to an additional 8 years (total of 58 years) show an increase in thickening at an increased rate, however there are limitations on research as to the speed and rate of thickening, encroachment, fuel assessments, hydrology and possible threat of increased anthropogenic fires.

Fire modelling is an important tool for land and fire managers, however the complexity of the natural system, multiple drivers of fire, climate and vegetation impede high resolution modelling. Extensive research and work has been undertaken on modelling in the USA and northern hemisphere, which has improved fire modelling projects substantially over the past decade. Further research into physiological parameters of different species of southern hemisphere plants would greatly improve model driven scenario outputs, especially for the iconic Fraser Island.

Future research into multiproxy fire ecology that employs analyses of charcoal, pollen geochemistry and radioactive isotopes to reconstruct fire, vegetation and sediment dynamics (Beck *et al.* 2017) is recommended. In addition research into fire intensity, severity and ecosystem response to changing fire frequency, fuel loads and distribution in a changing climate of this region is required.

The *E. minus* wetland, sclerophyll forest ecotone is dynamic in that fire acts as a system destabiliser preventing encroachment and thickening of the sclerophyll boundary vegetation into the *E. minus* wetlands. With climate change, increasing CO_2 levels, changes to temperature and precipitation, it is expected that there will be an increase in fire frequency and intensity, promoting vegetation thickening that will impact *E. minus* wetlands at Moon Point. Further research is needed on the relative influence of these factors and into fire as a driver in maintaining vegetation in a semi quasi climax state.

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Appendix 1.

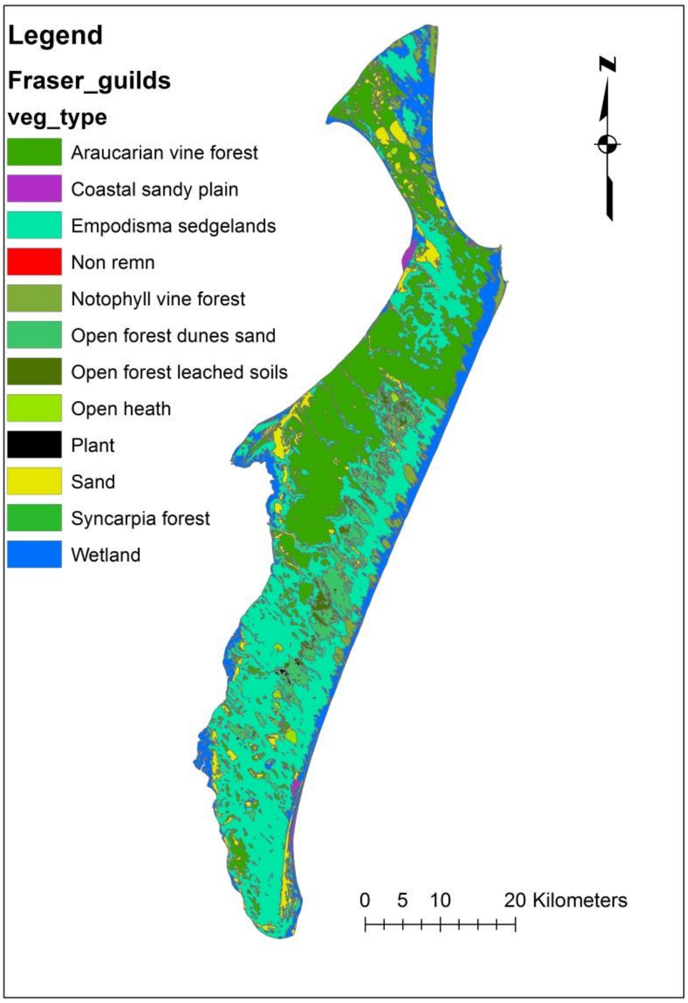
Modified Sampling Protocols for
FireBGCv2 – Great Sandy Region 2015

SAMPLING OBJECTIVES

The overall objective for our field sampling is to collect vegetation and fuels data for parameterisation of FirebGCv2, a fire-forest succession model. To this end, we will sample stands that represent the major vegetation types that comprise the simulation area of the Great Sandy Region, Queensland, Australia. In these stands we will sample vegetation (overstory and understory) and fuels within nested circular plots. Representative vegetation types are defined as combinations of dominant overstory vegetation and successional stage. Ideally, we will sample a minimum of 30 plots over our sampling campaign, and up to 50 as time allows.

SAMPLING LOCATION

We plan on opportunistically sampling the representative forest types in the simulation area defined in the map below



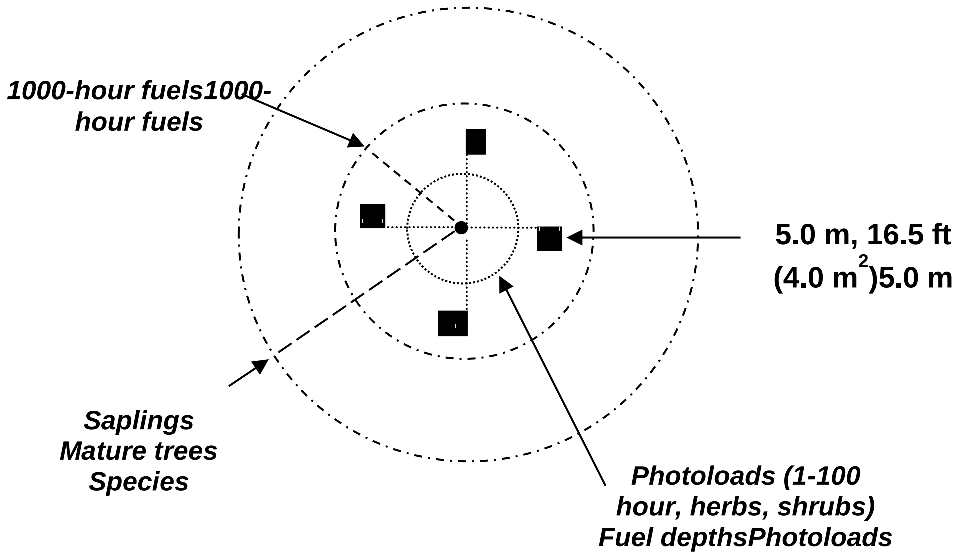
The entire island is the simulation area and the major vegetation types are shown by the different colors (codes as per RE BVG). We plan to sample those vegetation types and their structural stages (early, mid-, and late seral) both inside and outside the simulation area. We hope to have at least one plot for each vegetation type-structural stage combination.

GENERAL METHODS

Plots will be opportunistically located within representative stand types within the study area, defined as combinations of dominant overstory vegetation and successional stage. For example, stand cover types within Great Sandy Region are classified as combinations of dominant tree species within successional stages that represent time since last disturbance.

Within each plot we will collect data within nested, hierarchical plots as follows:

Data type		***Plot radius/area***
Trees	Seedlings only	3.6 m/.004 ha
	Saplings through Mature	11.34 m/.04 ha
Understory (herbs, forbs, grasses)		11.34 m/.04 ha
Fuels (Photoload sampling)	1-1000 hour	4 x 1 m^2 plots at 5 m
	Herb and forb biomass	4 x 1 m^2 plots at 5 m
	Fuel depths	2 corners in each 1 m^2 plot



Emergency information

Dial 000 for emergency, fire, medical or ranger assistance or to report accidents or injuries.

General information

1) Compass declination of 10.35˚ East
2) Datum and projection for GIS data: GDA94, Zone 56
3) Registration Code: RAL
4) Project Code: GSRBGC
5) All measurements are taken in centimetres and metres; GPS datum is GDA94
6) Collect all flagging and other field gear – leave no trace!!

SAMPLING STEPS

I. Find and Establish Plot

1) Look for a cover type not yet sampled on the Master Vegetation Worksheet (Appendix 3 - example). **Sample at least 3 plots per vegetation type combination for Priority 1 plots, 2 plots per vegetation type combination for Priority 2 plots, and 1 plot per vegetation type combination for Priority 3 plots.** Select plots such that a range of geographic, topographic, and edaphic settings will be sampled. For example, plots in the LP1 cover type should be located at sites with different slopes and aspects, and separated in space within the study area.

2) Look for a representative plot location within stand for that cover type:
   a. A valid plot location is located within a stand that is at least 2.0 km^2 in area.
   b. The stand should contain homogeneous vegetation conditions with respect to species composition and stand structure.
   c. Plot should be at least 150-ft from any major change in vegetation or ecosystem conditions (i.e. roadway, ecotone, water) and 150-ft from edge of the stand.
   d. Make sure plot doesn't contain atypical features (i.e. brush piles, trails, camp spots).
   e. Plot should be representative of vegetation conditions, including not just vegetation cover type but also plant health, topography (average elevation, aspect, slope), fuel, and disturbance history (fire, insect, disease).

3) Throw chaining pin over shoulder and establish plot center where pin lands.

4) Place microplot frame (1-m x 1-m PVC frame) at four locations around plot center: true north, 90˚, 180˚ and 270˚ azimuths, each at 16.4-ft (5-m) from plot center, to the right of each azimuth (**see plot schematic**). Use pinflags to mark plot areas and do not walk in these areas while sampling other elements. Microplots are for fuels and understory biomass sampling.

5) Identify plot. Plot number is 001 through 00x in sequence

6) Flag trees to indicate whether they are in or out of plot – "knot in is not in the plot." Plots are 0.04 ha circular plots with a 37.2 foot radius from plot center, adjusted for slope as in **slope correction table in Appendix 2**.

II. Plot Description form:

1) Plot info: units - E (English); sampling event: M (monitoring); plot radius – 37.2-ft
2) Georeferenced position: UTM S and E coordinates; Zone 56; Datum GDA94; error; Units (m)
3) Biophysical settings: elevation (from GPS); aspect - direction plot (as a whole) is facing; slope - average of uphill and downhill slope from plot center; landform – use **NRIS landform codes in Appendix 2**; Vertical and Horizontal slope shape – use **slope shape codes in Appendix 2.**
4) Tree, herb, ground, shrub cover – use **cover class codes in Appendix 2**.
   a. Vegetation cover does not need to sum to 100 percent by life form because there will probably be overlapping cover across all life forms.

However, the total cover for each life form must always be greater than any of the covers estimated for the size classes within that life form.

b. “Ground” cover must sum to 100%, as it includes bare ground, soil etc.

5) Composition: Upper Dom Spp (>10-ft tall); Mid Dom Spp (3 to 10-ft tall); Lower Dom Spp (< 3-ft) (dominant species must have ≥10% canopy cover); Potential Veg ID (habitat types, don’t need phase); Potential life form – use **potential life form codes in Appendix 2.**

6) Fuels: Surface fire behavior fuel model and fuel photo ID– use Andersons; stand height - highest tree stratum that contains ≥10% canopy cover; canopy fuel base height – average lowest height with sufficient canopy to propagate fire vertically into canopy; Canopy cover – percent canopy above 6-ft using **cover class codes in Appendix 2**.

7) Photo documentation: Photo 1 facing due north; Photo 2 facing due east. Capture plot center chaining pin plus representative plot vegetation in photos.

   a. Take additional photos for future use as cover type reference. Opportunistically take photos as representative sites are encountered. BE SURE TO RECORD PHOTO # and corresponding cover type.

III. Species composition form: record species composition within a macroplot with radius of 37.2 ft measured from plot center. **Note: Do not record or collect plant species that compose less than 5% of the macroplot area. This form is used to record both understory shrubs/herbaceous species AND SEEDLINGS.**

1) Item Code: all species will be identified; use 4-letter code (first two letters of genus plus first two letters of species). Retain a master list (**Appendix 5**) of codes that contain the full name (or add new species to the one provided). Also indicate on this list if you use a five letter code to distinguish between species with the same four letter code (e.g., *Populus tricocarpa* (POTRC) and *Populus tremuloides* (POTRM)). If you can't identify the species enter UNKN1, UNKN2, etc. and collect a voucher (reference) specimen for each species that includes diagnostic features such as flowers, seed heads, leaves, and stem. Label the specimen with the same number as the species UNKN# id, the plot number, and the date, and be sure to estimate cover for **each** of the unknown species.

2) Status: Live/ Dead

3) Size Class: Use **size class codes in Appendix 2**

4) Cover: Use **cover class codes in Appendix 2**; can have >100% cover due to overlapping layers.

5) Height: for herbs, measure only to point that includes about 80% of plant biomass (i.e. inflorescence height in graminoids is not measured).

IV. Fuels form:

1) 1-100 hour fuels: use photoload sequences to estimate loadings in each of four 1-m x 1-m plot frames
    a. Select two photoload pictures that bound observed loading in sample unit
    b. Compute loading estimate by interpolating between pictures
    c. Shrubs, Herbs: use photoload sequence to estimate loadings in each of four 1-m x 1-m plot frames
    d. As above, estimate loading
    e. Adjust loadings in the field for height difference relative to reference species, but also record average plant height.

2) Fuel depths: Measure fuel depth at points 4 meters (13.1-ft) and 5 meters (16.4-ft) from plot center within each microplot, along the trailing edge of the plot frame as you sweep clockwise around the pinwheel.

3) 1000-hr fuels:
    a. Measure all logs (>3-in) within 11.8-ft radius from plot center working clockwise (correct for slope as needed, using **slope correction table in Appendix 2**).
    b. Small and large end diameters: Use a ruler or calipers to measure the small and large ends of each log in centimetres. If part of the log has its central axis below the duff layer, measure the last point of the log above the duff layer.
    c. Length: Use a loggers tape to measure the length of each log in metres (to one decimal). Measure the length along the central axis, defined as a straight line connecting the two furthest points on a particle. For straight logs this will correspond with the pith, but for curved or branched particles the central axis may fall outside the particle boundary.
    d. Decay class: Record decay class for each log using **decay class codes in Appendix 2**.

V. Tree Data Sampling

1) **Measure seedlings (< 4.5-ft height)**
    a. Sample within 11.8-ft radius from plot center (correct for slope as needed, using **table 1**) working clockwise.
    b. Begin due north of plot center, tally seedlings by species into health status (Live or Dead) and height classes (see **height class table in Appendix 2**).

2) **Measure saplings (≥ 4.5-ft height & < 4-in DBH):** tally saplings in species-diameter class-status groups, rather than individuals.
    a. As above, start due north of plot center and work clockwise but within 18.5-ft.
    b. Diameter classes: use **diameter classes in Appendix 2**
    c. Health Status (Live/Dead)
    d. Average height of species-diameter class-status group.
    e. Average crown fuel base height

3) **Measure all mature trees (≥ 4.5-ft height and > 4-in DBH)**
    a. Begin due north of plot center and work around plot clockwise. Mark first tree with double flagging.
    b. Health: Live or Dead. Record all trees regardless of health status.
    c. DBH – measure on uphill side of tree (if on slope) and 4.5-ft above ground surface, to nearest 0.10 inch. Keep tape perpendicular to tree stem.

d. Height – measure height of first 3 trees (including crown base height) and estimate remainder.

  i. Go uphill or on flat a distance a little farther than tree is tall. Check percent scale from clinometer. IF > 100% then walk back farther.

  ii. Take percent scale at top and bottom of trees; subtract these readings (note, bottom reading is usually negative); Multiply by distance from tree; divide by 100: $(\%_{top} - \%_{bottom})$ * Distance/100

e. Crown base height: **for live and dead trees** - height of dead material sufficient to carry fire from lower to upper part of tree crown; or, if dead fuel insufficient, height of lowest live foliage.

f. Crown class: Use **crown class codes in Appendix 2**.

g. Mortality code: Identify the damage that initiated the tree mortality (the first damage to the tree causing its reduced vigor) using **mortality codes in Appendix 2. Pay special attention to signs of beetle kill – we'll want to know what proportion of trees in a stand have died from mountain pine beetle attack.**

h. Damage code: Identify the Enter the codes that describe evidence of a damaging agent on the tree using **damage codes in Appendix 2. Pay special attention to signs of beetle kill – we'll want to know what proportion of trees in a stand have been attacked by mountain pine beetles.**

<u>When in doubt about sampling protocols or methods consult the FIREMON GTR.</u>

TREE DATA (TD) FIELD DESCRIPTIONS

Field 1: **Macroplot Size**. Macroplot size (acre/ha).
Field 2: **Microplot Size**. The size of the microplot where trees less the 4.5 ft tall are sampled (acre/ha).
Field 3: **Snag plot size**. The size of the plot where dead trees greater than breakpoint diameter are measured (acre/ha).
Field 4: **Breakpoint Diameter**. DBH above which trees are measured individually and below which trees are tallied by diameter-species-status classes (inch/cm).
Table 1: Mature Trees. Trees greater than breakpoint diameter at breast height
Field 5: **Tag Number.** Tag number attached to mature trees. The tagged numbers need not be in sequence.
Field 6: **Species Code.** Either the NRCS plants species code or the local code for that species. Precision: No error.
Field 7: **Tree Status.** Tree status code that best describes the current health of the tree. Precision: ±1 class.
H—Healthy tree with little biotic or abiotic damage.
U—Unhealthy tree with some biotic or abiotic damage, and this damage will reduce growth. However, it appears the tree will not immediately die from the damage.
S—Sick tree with extensive biotic or abiotic damage, and this damage will ultimately cause death within the next 5 to 10 years.
D—Dead tree or snag with no living tissue visible.
Field 8: **DBH**. The diameter of the tree at breast height (inch/cm). Precision: ±0.1 inch/0.25 cm.
Field 9: **Tree Height**. The vertical height of the tree (ft/m). Precision: ±1 ft/0.3 m.
Field 10: **Live crown percent.** The percent class that best describes the percent of the tree stem that is supporting live crown based on the distance from the ground to the top of the live foliage. Valid classes are in table TD-5 of the TD sampling method. Precision: ±1 class.
Field 11: **Crown Fuel Base Height.** Height above the ground of the lowest live and/or dead fuels that have the ability to move fire higher in the tree (ft/m). Precision: ±1 ft/0.3 m.
Field 12: **Crown Class.** Code that describes the tree crown's position in forest canopy. Precision: ±1 class.
O—Open grown, the tree is not taller than other trees in the stand but still receives light from all directions.
E—Emergent, the crown is totally above the canopy of the stand.
D—Dominant, the crown receives light from at least three to four directions.
C—Codominant, the crown receives light from at least one to two directions.
I—Intermediate, the crown only receives light from the top.
S—Suppressed, the crown is entirely shaded and underneath the stand canopy.
Field 13: **Tree Age.** Tree age taken from sample core. Precision: ±10 percent of total years.
Field 14: **Growth Rate.** The distance measured across the 10 most recent growth rings (inch/mm). Precision: ±0.01 inch/0.1 mm.
Field 15: **Decay Class**. Decay measure of snags. Valid classes are in table TD-7 of the sampling method. Precision: ±1 class.
Field 16: **Mortality Code**. Damage that initiated the tree mortality (the first damage to the tree causing its reduced vigor). Valid codes are in table TD-8 of the sampling method. Precision: best guess.
Field 17 to Field 20: **Damage and Severity Codes**. Enter the codes that describe evidence of a damaging agent on the tree and the severity of the damage in order of prevalence on the tree. See **Appendix A: NRIS Damage Categories, Agents, Severity Ratings, and Tree Parts**. Precision: Correct damage category, ±1 severity class.
Field 21: **Bole Char Height**. Enter the height of the highest contiguous char measured on the downhill side of the tree. Precision: ±0.1 ft/0.3 m.
Field 22: **Percent Crown Scorched**. Enter the percent of crown that has been killed by fire. Include both scorched and consumed foliage. Valid codes are in table TD-5 of the sampling method. Precision: ±1 class.
Field 23: **Local Variable**. User defined value or code.
Table 2: Saplings—Trees less than breakpoint diameter and taller than 4.5 ft.
Field 24: **Diameter Class**. Class of the trees being sampled. The Analysis Tools program assumes that the diameter class value in this field represents the midpoint of the DBH range of the trees being sampled. Precision: No error
Field 25: **Species Code**. Code of sampled entity. Either the NRCS plants species code or the local code for that species. Precision: No error.
Field 26: **Status Code**. Tree status code that best describes the current health of the tree. Codes presented above in the Field 7 description. Precision: ±1 class.
Field 27: **Count**. The number of trees tallied for the appropriate diameter-species-status class. Precision: ±10 percent of total count.
Field 28: **Average Height**. The average height of all trees tallied for this diameter-species-status class. Precision: ±1 ft/0.3 m.
Field 29: **Average Live crown percent**. Enter the average live crown percent of the trees tallied for this diameter-species-status class. Valid classes are in table TD-5 of the TD sampling method. Precision: ±1 class.
Field 30: **Local Code**. User defined value or code.
Table 3: Seedlings—Trees less than 4.5 ft tall.
Field 31: **Height Class**. Class of the trees being sampled. The Analysis Tools program assumes that the height class value in this field represents the midpoint of the height range of the trees being sampled. Precision: No error.
Field 32: **Species Code.** Code of sampled entity. Either the NRCS plants species code or the local code for that species. Precision: No error.
Field 33: **Status Code**. Tree status code that best describes the current health of the tree. Codes presented above in the Field 7 description. Precision: ±1 class.
Field 34: **Count.** The number of trees tallied for the appropriate height-species-status class. Precision: ±10 percent of total count.
Field 35: **Local Code**. User defined value or code.

Cover type classifications

Cover Type	*Trees*	*Understory*	*Age*	*Disturbance*	*Fuels*
LP0	**Potential** LP seedlings	Strawberries Aster Elk sedge	0-40	Recent	Forbs Grasses Rotten logs
LP1	Dense LP saplings & poles	Sparse vegetation Rotten logs	50-150		Live needles Rotten logs
LP2	Closed canopy LP	ES seedlings < 8ft tall SAF seedlings < 8ft tall Thick herbaceous vegetation Grouse whortleberry	150-300		Generally too wet Possible beetle kill
LP3	Mixed conifer LP, ES, SAF, WBP	Spruce and fir Seedlings to saplings Globe huckleberry	> 300	Most Common fire origin	CONTINUOUS deep litter/duff rotten logs
LP (No Spruce/Fir)	LP climax Some WBP possible	LP seedlings & saplings Scattered herbs & small shrubs	> 300	Rare, spotty	Sparse
SF (Spruce/Fir)	ES& SAF climax LP or WBP in areas	*** SF0-SF1-SF2-SF progression possible on very wet or cold sites****			Abundant grasses, herbs and shrubs
WB0	Post fire WBP	Abundant, very green		Recent	
WB1	Even-aged WBP younger than neighboring stands	*****similar to LP1****		Rare	
WB2	*****similar to LP2	*****		Rare	High elevation
WB3	Seral WBP Climax ES,SAF,LP	Dense spruce and fir		Infrequent	Generally too wet
DF0		Dense DF seedlings		Recent	Grasses & shrubs
DF1	Even-aged DF younger than neighboring stands				Dead branches Continuous ladder fuels
DF2	** similar to LP2***	Abundant undergrowth eg. Pinegrass, snowberry, ninebark			
DF	Dominated by DF some LP	Pinegrass, snowberry, ninebark			Needle litter Surface fuels Possible doghair

Sampling measurement classes and codes (from FIREMON)

Plot Description (PD) Form:

NRIS Landform Codes

Landform	Code
GMF	Glaciated mountains-foothills
UMF	Unglaciated mountains-foothills
BRK	Breaklands-river breaks-badlands
PLA	Plains-rolling planes-plains w/breaks
VAL	Valleys-swales-draws
HIL	Hill-low ridges-benches
X	Did not assess

Cover class codes

Code	Cover Class
0	Zero
0.5	>0 – 1
3	>1 – 5
10	>5 – 15
20	>15 – 25
30	>25 – 35
40	>35 – 45
50	>45 – 55
60	>55 – 65
70	>65 – 75
80	>75 – 85
90	>85 – 95
98	>95 - 100

Slope shapes

Code	Slope Shape
LI	Linear or planar
CC	Depression or concave
PA	Patterned
CV	Rounded or convex
FL	Flat
BR	Broken
UN	Undulating
OO	Other shape
X	Did not assess

Potential life form codes

Code	Potential life form
AQ	Aquatic—Lake, pond, bog, river
NV	Nonvegetated—Bare soil, rock, dunes, scree, talus
CF	Coniferous upland forest—Pine, spruce, hemlock
CW	Coniferous wetland or riparian forest—Spruce, larch
BF	Broadleaf upland forest—Oak, beech, birch
BW	Broadleaf wetland or riparian forest—Tupelo, cypress
SA	Shrub dominated alpine—Willow
SU	Shrub dominated upland—Sagebrush, bitterbrush
SW	Shrub dominated wetland or riparian—Willow
HA	Herbaceous dominated alpine—Dryas
HU	Herbaceous dominated upland—grasslands, bunchgrass
HW	Herbaceous dominated wetland or riparian—ferns
ML	Moss or lichen dominated upland or wetland
OT	Other potential vegetation
X	Did not assess

Species Composition (SC) Form:

Tree size class		
Codes	**English**	**Metric**
TO	Total cover	Total cover
SE	Seedling (<1 inch DBH or <4.5 ft height)	Seedling (<2.5 cm DBH or <1.5 m height)
SA	Sapling (1.0 inch–< 5.0 in. DBH)	Sapling (2.5–<12.5 cm DBH)
PT	Pole tree (5.0 inches–<9.0 in. DBH)	Pole tree (12.5–<25 cm DBH)
MT	Medium tree (9.0 inches–<21.0 in. DBH)	Medium tree (25–<50 cm DBH)
LT	Large tree (21.0 inches–<33.0 in. DBH)	Large tree (50–<80 cm DBH)
VT	Very large tree (33.0+ inches DBH)	Very large tree (80+ cm DBH)
NA	Not applicable	Not applicable

Cover class codes

Code	Cover Class
0	Zero
0.5	>0 – 1
3	>1 – 5
10	>5 – 15
20	>15 – 25
30	>25 – 35
40	>35 – 45
50	>45 – 55
60	>55 – 65
70	>65 – 75
80	>75 – 85
90	>85 – 95
98	>95 - 100

Fuel Loading (FL) Form:

Shrub/herb size class		
Codes	**English**	**Metric**
TO	Total cover	Total cover
SM	Small (<0.5 ft height)	Small (<0.15 m height)
LW	Low (0.5–<1.5 ft height)	Low (0.15–<0.5m height)
MD	Medium (1.5–<4.5 ft height)	Medium (0.5–<1.5 m height)
TL	Tall (4.5–<8 ft height)	Tall (1.5–<2.5 m height)
VT	Very tall (8+ ft height)	Very tall (2.5+ m height)
NA	Not applicable	Not applicable

Correct for slope by multiplying distanc
correction factor (CF). Original plot size

Slope	CF	New radius (ft)	Slope	C
10	1.00	37.2	100	1
20	1.02	37.9	110	1
30	1.04	38.7	120	1
40	1.08	40.2	130	1
50	1.12	41.7	140	1
60	1.17	43.5	150	1
70	1.22	45.4	160	1
80	1.28	47.6	170	1
90	1.35	50.2	180	2

Decay class codes for 1000-hr fuels

Decay class	Description
1	All bark is intact. All but the smallest twigs are present. Old needles probably still present. Hard when kicked.
2	Some bark is missing, as are many of the smaller branches. No old needles still on branches. Hard when kicked.
3	Most of the bark is missing, and most of the branches less than 1 inch in diameter also missing. Still hard when kicked.
4	Looks like a class 3 log but the sapwood is rotten. Sounds hollow when kicked, and you can probably remove wood from the outside with your boot. Pronounced sagging if suspended for even moderate distances.
5	Entire log is in contact with the ground. Easy to kick apart but most of the piece is above the general level of the adjacent ground. If the central axis of the piece lies in or below the duff layer then it should not be included in the CWD sampling, as these pieces act more like duff than wood when burned.

Tree Data (TD) Form: Mortality codes for mature trees

Table TD-8—Use these mortality codes to identify the primary (first) cause that killed the tree.

Mortality code	Description
I	Insect caused
D	Disease caused
A	Abiotic (flooding, erosion)
H	Harvest caused
U	Unable to determine
X	Did not assess

Crown classes

O – Open grown: tree not near any other tree
E – Emergent: crown is totally above canopy of stand
D – Dominant: crown receives light from at least 3-4 directions
C – Codominant: crown receives light from at least 1-2 directions
I – Intermediate: crown only receives light from the top
S – Suppressed: crown is entirely shaded and underneath stand canopy.

Height classes for seedlings

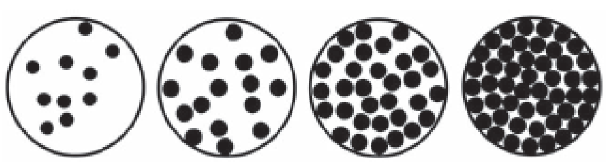
Class	Height range (ft)
0.2	>0 – 0.5
1	>0.5 – 1.5
2	>1.5 – 2.5
3	>2.5 – 3.5
4	>3.5 – 4.5

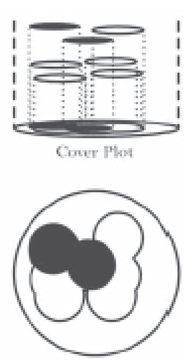
Class	Diameter range
0.5	>0-1
1.5	1-2
2.5	2-3
3.5	3-4

Damage and severity codes—short list

Damage code	Description	Severity code
00000	No damage	No damage
10000	General insects	101—Minor—Bottlebrush or shortened leaders or <20% of branches affected, 0—2 forks on stem or <50% of the bole with larval galleries 102—Severe—3 or more forks on bole, or 20% or more branches affected or terminal leader dead or 50% or more on the bole with visible larval galleries.
19000	General diseases	191—Minor—Short-term tree vigor probably not affected. 192—Severe—Tree vigor negatively impacted in the short term.
25000	Foliage diseases	251—Minor—<20% of the foliage affected or <20% of crown in brooms. 252—Severe—>20% of the foliage affected or >20% of the crown in brooms.
50000	Abiotic damage	501—Minor—<20% of the crown affected, bole damage is <50% circumference. 502—Severe—>20% of the crown affected bole damage is >50% circumference.
90000	Unknown	900—0–9% affected 901—10–19% affected 902—20–29% affected 903—30–39% affected 904—40–49% affected 905—50–59% affected 906—60–69% affected 907—70–79% affected 908—80–89% affected 909—90–99% affected

Estimating cover:





Master Vegetation Worksheet

COVER TYPE	AREA (km^2)	% STUDY AREA	CANCOV	FM	CBH	TREEHT	CBD	Plot IDs
lp0	448.76	25.61	1	5	1	10	10	
lp2	413.49	23.60	3	8	104	185	22	
lp	200.84	11.46	1	8	104	185	9	
nf	165.33	9.43	0	2	0	0	0	
lp1	131.69	7.52	3	8	99	147	22	
lp/lp2	112.89	6.44	2	8	104	185	15	
lp3	88.22	5.03	3	10	12	204	26	
lp1/lp	81.35	4.64	1	8	99	185	16	
df	17.58	1.00	1	8	30	223	10	
wb	12.39	0.71	2	8	24	167	14	
wb/nf	9.82	0.56	1	2	24	167	9	
sf	7.49	0.43	3	10	13	186	26	
df0	5.73	0.33	1	1	0	0	0	
lp/lp3	4.92	0.28	2	10	12	204	15	
wb0	4.36	0.25	1	1	0	0	0	
wb/df	3.90	0.22	2	8	24	223	16	
lp2/nf	2.93	0.17	1	2	104	185	9	
sf/nf	2.70	0.15	1	2	13	186	10	
lp3/nf	1.29	0.07	1	2	12	204	10	
df/nf	0.38	0.02	1	2	30	223	10	
asp	0.29	0.02	1	9	90	180	5	
lp0/nf	0.11	0.01	1	1	1	10	10	
water	35.85	2.05	0	98	0	0	0	
	1752.31							
CANCOV = Canopy cover								
FM = FPBS fuel models 1-10								
CBH = Crown base height, meters *10								
TREEHT = Tree height, meters *10								
CBD = Crown bulk density, kg/m-3 *100								

Equipment needs for Forest Stand Inventory sampling: Plot Description, Fuel Load, Tree Data, and Species Comp.

Method	Equipment
Plot Description (2 sets)	Maps, charts and directions Pin to mark plot center Camera with flash and batteries GPS (1 main plus one backup) Clear plastic ruler Clinometer Clipboard Cloth tape Compass Flagging Indelible ink pen (Sharpie) Lead pencils with refills Map protector or plastic bag Plot sheet protector or plastic bag Logger's tape Pocket calculator Field notebook PD data forms and cheat sheet and plot sheet protector
Fuel Load (2 sets)	Go/No-Go gauge 1-m by 1-m PVC plot frame Sharp trowel Photoload guide (2) FIREMON cheatsheets, fuels data forms
Tree Data	Hard hats TD data forms Laser hypsometer
Species Composition	Plant press Newspaper Masking tape Flagging PD data forms and plot sheet protector
Safety	First aid kit(s) 2-way radios Bear spray
Other Items	Laptop Handheld GIS unit Extra batteries for camera, GPS, 2-way radios Duct tape

Code	Latin name	Common name	Code	Latin name	Common name