Report
NR001 | **2012**

Vegetation–fire interactions in central arid Australia

Towards a conceptual framework

JB Marsden-Smedley
D Albrecht
GE Allan
C Brock
A Duguid
M Friedel
AM Gill
KJ King
J Morse
B Ostendorf
D Turner

NINTI ONE

Ninti One Limited Research Report NR001

ISBN: 978-1-74158-211-6

Contributing author affiliations

J.B. Marsden-Smedley[a,g,h,i], D. Albrecht[b], G.E. Allan[c,h], C. Brock[b,j], A. Duguid[b,h], M. Friedel[d], A.M. Gill[a,g], K.J. King[a,g], J. Morse[e], B. Ostendorf[f] and D. Turner[f,h]

[a]Fenner School of Environment and Society, The Australian National University, Canberra, Australian Capital Territory 2600, Australia. malcolm.gill@anu.edu.au; karen.king@anu.edu.au

[b]Department of Natural Resources, Environment, the Arts and Sport, PO Box 1120, Alice Springs, Northern Territory 0871, Australia. dave.albrecht@nt.gov.au; angus.duguid@nt.gov.au

[c]Bushfires NT, PO Box 2533, Alice Springs, Northern Territory 0871, Australia. grant.allan@nt.gov.au

[d]CSIRO Ecosystem Sciences, PO Box 2111, Alice Springs, Northern Territory 0871, Australia. margaret.friedel@csiro.au

[e]PO Box 7245, Tathra, New South Wales 2550, Australia. morse2@iinet.net.au

[f]School of Earth and Environmental Sciences, University of Adelaide, PMB 1, Glen Osmond, South Australia 5064, Australia. bertram.ostendorf@adelaide.edu.au; dorothy.turner@adelaide.edu.au

[g]Bushfire Cooperative Research Centre, Level 5, 340 Albert Street, East Melbourne, Victoria 3002, Australia

[h]Desert Knowledge Cooperative Research Centre, Alice Springs, Northern Territory 0870, Australia.

[i]Present address: School of Geography and Environmental Studies, University of Tasmania, Sandy Bay, Tasmania 7005, Australia. Jon.MarsdenSmedley@utas.edu.au

[j]Present address: PO Box 411, Yarra Junction, Victoria 3797, Australia. chris@brockenvironmental.com.au

Corresponding author: Margaret Friedel, CSIRO Ecosystem Sciences, PO Box 2111, Alice Springs, Northern Territory 0871, Australia; Tel: +61 8 8950 7140; Fax: +61 8 8950 7187; Email address: margaret.friedel@csiro.au

Citation

Marsden-Smedley JB, Albrecht D, Allan GE, Brock C, Duguid A, Friedel M, Gill AM, King KJ, Morse J, Ostendorf B and Turner D. 2012. *Vegetation–fire interactions in central arid Australia: towards a conceptual framework*. Ninti One Research Report NR001. Ninti One Limited. Alice Springs.

Acknowledgement

The work reported in this publication was supported by funding from the Australian Government Cooperative Research Centres Program through the Desert Knowledge CRC and the Bushfire CRC; the views expressed herein do not necessarily represent the views of these CRCs or their participants. This paper is an output of the Desert Fire project of the Desert Knowledge CRC, and both the Bushfire and Desert Knowledge CRCs provided funding for the development of this paper. We thank Peter Latz for his very important contributions to the workshop. The assistance of the Alice Springs Desert Park in donating the use of the workshop venue is acknowledged and appreciated. We also acknowledge the organisations that supported the participation of their staff in the workshop and in the development of this paper, and we thank Dr Ross Bradstock and Dr Glenn Edwards for their review of an earlier draft of this report.

For additional information please contact

Ninti One Limited
Communications Manager
PO Box 154, Kent Town, SA 5071, Australia

Telephone +61 8 8959 6000 Fax +61 8 8959 6048 www.nintione.com.au

Vegetation–fire interactions in central arid Australia: towards a conceptual framework

JB Marsden-Smedley

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Abstract

A comprehensive understanding of vegetation and fire dynamics and their interactions is important for effective land management in central arid Australia, the majority of which is vegetated with natural and semi-natural (grazed) vegetation. The potential loss of fire-sensitive populations of tree and shrub species and the impact of invasive grasses on fire regimes are of concern. This paper synthesises current knowledge, from published literature and expert opinion, about vegetation–fire interactions in central arid Australia. An initial conceptual framework for vegetation–fire interactions is presented for exploring the relationship between average fire frequency and potential fuel load in different vegetation types. This framework is intended to encourage debate and stimulate further research regarding these issues. The paper also considers the influence of both fire management and other drivers of changed fire regimes, such as climate change and the spread of introduced pastoral grasses, in particular buffel grass (*Cenchrus ciliaris*).

Key words: fire, spinifex, mulga, ecological succession, climate change, buffel grass

1. Introduction

Fire is a fundamental component of the ecology of arid Australia. Fire regimes anywhere are a product of interactions among several factors, typically, vegetation type (fuel characteristics and flammability), terrain steepness, land use, ignition sources and weather. The long-term ecological effects of fire are strongly related to fire regimes as well as to significant individual fire events, and they are the basis of many fire effects in the long term (see Bradstock et al. 2002, Gill 2008). In arid Australia rainfall is probably the most important driver of fuel loading and, consequently, of fire regimes. Extensive wildfires (or ‘unplanned fires’) follow several years of above-average rainfall (Edwards et al. 2008; Griffin et al. 1983; Turner et al. 2008, 2011). This contrasts to wildfires in forested regions, which correlate strongly with droughts (Luke & McArthur 1978). Human activity is another major influence on fire regimes through the deliberate and accidental lighting of fires, modification of fuel loads and through longer-term changes such as species introductions, land degradation and infrastructure development. If fire is to be effectively managed in arid Australia, the influences of these factors need to be clearly understood.

Arid Australia contains a unique mosaic of vegetation types with different susceptibilities to fire. Large areas are characterised by hardy perennial grasses, known collectively as spinifex (*Triodia* spp.), that are highly fire-prone. Shrublands and low woodlands without spinifex also occupy large areas, often dominated by mulga (*Acacia aneura* F.Muell. ex Benth.) and are generally less fire prone (Edwards et al. 2008).

Fire regimes in central arid Australia have been influenced by lightning-induced fires, traditional Aboriginal burning practices and later by European land use. After the arrival of Europeans in central Australia in the 1860s, major changes to fire regimes were initiated by displacement of Aboriginal people and their land uses, and by introduced herbivores. More recently, the increasing extent and density of buffel grass (*Cenchrus ciliaris* L.), an introduced pasture grass, have arguably had a growing influence on the number, intensity and extent of wildfires in affected areas (e.g. Edwards et al. 2008). Climate change also has the potential to dramatically alter fire regimes (e.g. Cary 2002).

There are multiple land use and value systems in arid Australia. These include pastoralism, activities of Aboriginal people, conservation, tourism, mining and horticulture (e.g. Edwards et al. 2008). The majority of pastoralists regard fire as a threat to standing fodder and focus efforts on wildfire suppression (Allan & Tschirner 2009). A minority of pastoralists use fire proactively as a tool for promoting pasture production (e.g. Purvis 1986). Some land managers also use fire to reduce fuel loads in order to minimise the number and extent of unplanned fires and to protect assets. Some deliberate burning is specifically aimed at protecting biodiversity values, or to maintain Aboriginal resources and values (e.g. Bird et al. 2005, Burrows & Christensen 1990, Edwards et al. 2008, Gabrys & Vaarzon-Morel 2009, Griffin & Friedel 1985, Preece et al. 1989, Saxon & Allan 1984), although outcomes are not always predictable. A review of systems for planned burning in the south of the region was undertaken by Marsden-Smedley (2011) with a view to increasing the area subjected to active fire management.

The dearth of information on the complex interactions between vegetation, climate, land use and fire in arid Australia has been identified since the 1970s as a critical knowledge gap for land managers attempting to develop and apply appropriate management prescriptions for biodiversity conservation (Duguid et al. 2009; Myers et al. 2004, 2005; Preece 1990). A number of expert observers, including authors of this paper, have expressed concern about the implications of potential long-term change in vegetation and animal communities as a consequence of altered fire regimes. For example, an increase in invasive buffel grass within alluvial drainage lines seems likely to precipitate the decline in old river red gums (*Eucalyptus camaldulensis* Dehnh.) through increasing fire intensity and frequency, and thus the loss of tree hollows for dependent fauna. We propose that a conceptual framework for vegetation–fire interactions in arid Australia will help to reduce this knowledge gap.

This paper focuses on fire and vegetation dynamics; it combines a review of published scientific literature to 2011 with unpublished experiential knowledge gathered and analysed at a four-day scientific workshop held in Alice Springs in November 2005. Workshop participants are the authors of this paper. During the workshop we summarised the interactions among rainfall, terrain and soils, production, consumption and decay of biomass, introduced pastoral grasses and pastoral activities. We then developed a systematic description and schematic diagram of key vegetation types and their associated ecological characters. This conceptual framework is used here as a basis for discussing the implications of changes in fire regime, including those that may result from climate change or invasion of introduced grasses. We suggest that the conceptual framework and associated description of vegetation types should lead to more informed application of fire as a management tool and stimulate further research, including quantitative hypothesis testing, which was beyond the scope of the workshop.

The principal aims of this paper are to:

- define the main vegetation associations relevant to fire regimes in the central portion of the Australian arid zone
- synthesise current fire regime knowledge for key vegetation types
- develop a preliminary conceptual framework for vegetation–fire interactions.

2. Study area

Approximately 46% or 3.5 million square kilometres of Australia averages less than 350 mm of rain per year and is classified as arid (Bureau of Meteorology 2012a). Within this area, the primary focus for the workshop was the southern Northern Territory, or ‘central Australia’. However, many of the vegetation types discussed at the workshop have a wider distribution across the Australian arid zone, so we use the term ‘central arid Australia’ to describe our study area. We used the Interim Biogeographic Regionalisation for Australia (IBRA) version 6.1 (Australian Government Department of Environment and Heritage 2005) to identify areas with broadly similar landscapes to those of the arid southern part of the Northern Territory (Figure 1). This area comprises twelve IBRA bioregions, which have a combined area of 2 097 216 km^2 or about 27% of the land area of Australia (Table 1). The area covered by these IBRA bioregions is broadly similar to that classified as arid by the Bureau of Meteorology (2012a), and is within the central arid and southern arid rainfall regions derived by Russell-Smith et al. (2007) to describe the seasonal distribution of fires in Australia.

Central arid Australia has a very low human population density. The majority of residents live in a few communities and towns. By far the largest town within the area is Alice Springs, which has a population of about 25 000 (see Brown et al. 2008).

The landscapes of central arid Australia are dominated by broad flat expanses of sand plains, sand dunes, red earth plains and gravelly rises, with sparsely distributed salt lakes, rocky hills and mountain ranges. Most of the study area has highly weathered soils that are low in nutrients, particularly nitrogen and phosphorus (Litchfield 1962).

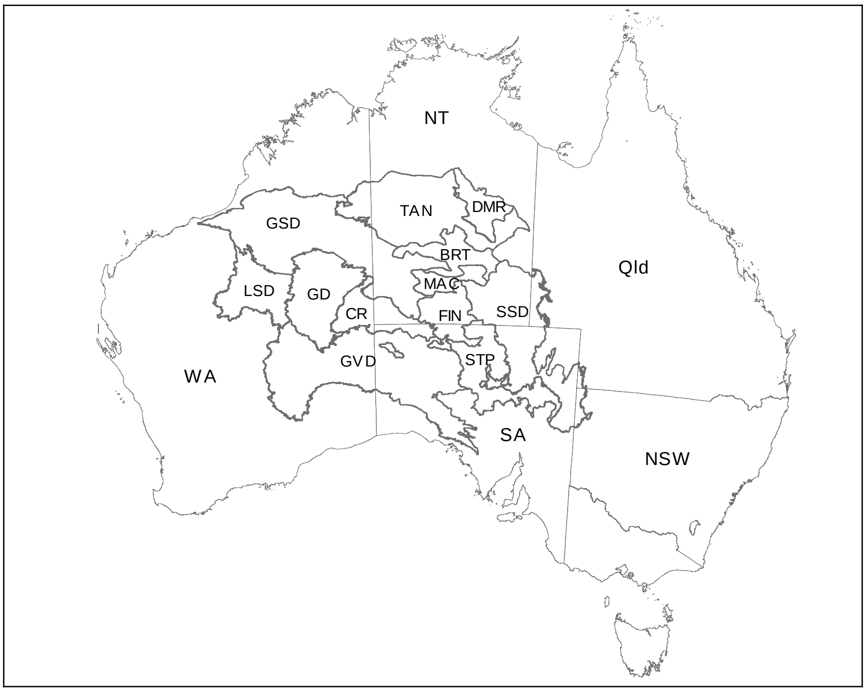

Figure 1: Biogeographic regions within the area covered by this paper

Notes: See Table 1 for region codes, based on the Interim Biogeographic Regionalisation for Australia, Version 6.1 (Australian Government Department of Environment and Heritage 2005). The regions are located in states and territories as follows: WA = Western Australia, NT = Northern Territory, SA = South Australia, Qld = Queensland, NSW = New South Wales.

Table 1: Biogeographic regions within the area covered by this paper

Bioregion	**Code**	**Area (km^2)**
Burt Plain	BRT	73 984
Davenport Murchison Ranges	DMR	58 220
Finke	FIN	73 943
Central Ranges	CR	101 375
Gibson Desert	GD	156 636
Great Sandy Desert	GSD	396 284
Great Victoria Desert	GVD	419 356
Little Sandy Desert	LSD	111 154
MacDonnell Ranges	MAC	39 385
Simpson Strzelecki Dunefields	SSD	273 405
Stony Plains	STP	132 728
Tanami	TAN	260 723
Total		**2 097 216**

Source: based on the Interim Biogeographic Regionalisation for Australia, Version 6.1 (Australian Government Department of Environment and Heritage 2005).

Within the context of global arid zones, much of the study area has a relatively high annual rainfall. Nevertheless, organisms must be highly adapted to aridity due to the high average temperatures, typically low humidity and the unreliable and episodic nature of rainfall (Newsome et al. 1996, Stafford Smith & Morton 1990). Climate varies substantially across the study area with a general trend to declining annual rainfall from north to south, with the southeast corner being the driest (Slatyer 1962).

Although Alice Springs is not necessarily representative of the range of climates across the area, it is the location with the longest record of weather observations, and these are summarised here from Bureau of Meteorology records. Summers are hot, with average minimum and maximum temperatures for December, January and February of about 20°C and 35°C respectively. Winters are mild, with average minimum and maximum temperatures for June–July of about 5°C and 20°C respectively. The average annual rainfall (calculated July to June due to the majority of the rainfall occurring over the warmer months) is 285 mm yr^{-1} (Figure 2). Rainfall is summer-dominated in Alice Springs and to the north, while to the south there is an increasing proportion of winter rainfall and the total average rainfall decreases. The average annual potential evaporation in Alice Springs is about 3175 mm (Bureau of Meteorology 2012b). Periods of several consecutive months with minimal or no rainfall occur in most years and at any time of year. More extended periods of below-average rainfall are also a recurring feature. Conversely, annual rainfall is often concentrated into a relatively small number of rain events, and years of rainfall well above average are not uncommon.

For the purposes of this paper, ‘major droughts’ have been defined as periods of at least six consecutive years when the rainfall averaged 81% or less than the long-term average (analyses of GE Allan). ‘Extreme wet periods’ are defined as three consecutive years or more when the rainfall averaged 118% or more than the long-term average. Accordingly, for the period 1874/75 to 2010/11 there were six major droughts in Alice Springs, varying in length from 6 to 12 years (i.e. 1895–1902, 1912–1918, 1925–1937, 1958–1966, 1988–1994 and 2002–2009). There were five extreme wet periods, of three to six years’ duration (i.e. 1937–1940, 1945–1949, 1966–1969, 1973–1979 and 1999–2002); 1920/21 was a single extremely wet year (1017 mm) as shown in Figure 2.

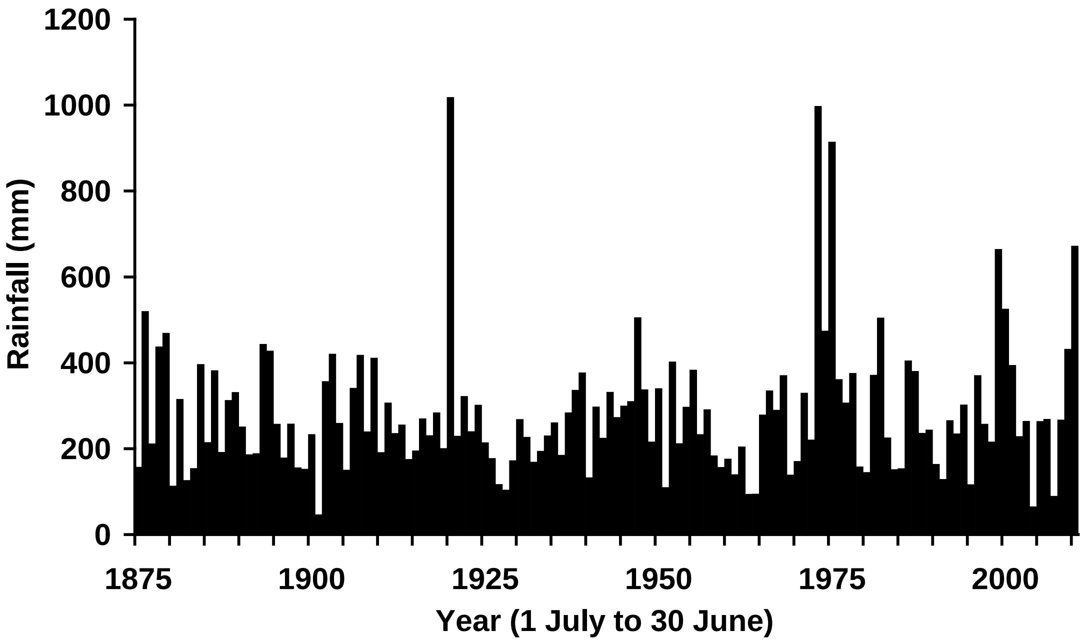

Figure 2: Annual rainfall between 1874/75 and 2010/11 at Alice Springs

Source: online data from Bureau of Meteorology 2012c

3. Vegetation patterns and processes

The ecology of individual plant species in central arid Australia is less thoroughly documented than for many parts of the world. However, general patterns and processes that strongly influence vegetation and fire are well understood. The distribution of plant species and therefore vegetation types is strongly influenced by the availability of two predominantly scarce resources: primarily soil water and secondarily soil nutrients. Consequently, vegetation structure and floristics are also influenced by redistribution of rainfall from elevated (run-off) areas to lower lying (run-on) areas (Perry & Lazarides 1962, Stafford Smith & Morton 1990).

Apart from a few species restricted to permanent waters, such as springs, all plant species require mechanisms for surviving extended dry periods that can range from several months without rain to severe droughts where minimal rain may fall for several years. Many species are ‘drought avoiders’ and survive dry times as seed. By contrast, many perennial plants have varying degrees of ‘drought tolerance’, and they include very long-lived and correspondingly drought-tolerant woody plants. The riparian areas of usually-dry river courses generally support relatively dense, tall vegetation dominated by river red gums (15 m or so high) which probably depend most commonly on unsaturated soil moisture that is replenished by river flows. Most water tables are too deep for plants to access.

Germination and establishment of most plant species, both perennials and annuals, are dependent on substantial rainfall events. Various factors influence which species germinate and survive and also their abundance and rate of growth. These include the season; amount and duration of the initial rain event; the occurrence of ‘follow-up rains’; temperature; humidity; soil moisture holding capacity; and position in the landscape, whether run-off or run-on. As a consequence, changes in vegetation composition after rainfall are somewhat unpredictable. However, in the warmer months, grasses tend to dominate new germination, whereas many forb species require cool season rain events to establish (e.g. Albrecht & Pitts 2004).

A description of these main ‘vegetation-habitat types’ that strongly influence, and are in turn influenced by, fire regimes was a main product of the workshop that formed the basis for this paper. We use the term ‘vegetation-habitat type’ because some vegetation types are defined purely on vegetation characteristics (structural and floristic), while others are also distinguished by terrain and soil characteristics. These types are described in Table 2, with summary information relating to their role in landscape fire patterns: landform, proportion of regional area occupied, soil nutrient availability, typical fuel type, average fire frequency, value to pastoralists (equating to rate of fuel removal by livestock), potential for invasion by buffel grass, potential for change promoted by spinifex and rate of fuel removal by termites.

Our descriptions of the vegetation-habitat types are based on various publications, together with our own observations of vegetation patterns, fire behaviour and species responses to fire. Although these types have been identified primarily for the arid south of the Northern Territory, they are considered to be indicative of patterns and processes across all of central arid Australia. The resolution of currently available vegetation maps is generally too coarse for mapping the vegetation-habitat types of the study area.

Existing vegetation descriptions and maps that influenced our description of vegetation-habitat types include a foundational study of vegetation and landscape in central arid Australia, undertaken in the 1950s (Perry 1962). Later, the vegetation of the entire Northern Territory was mapped at a very broad scale by Wilson et al. (1990). Both of these publications provide information on broad landscape characteristics but do not map, and only partially describe, distinct vegetation types at a scale relevant for understanding fire regimes. More detailed descriptions of vegetation are available for the southern Northern Territory (White et al. 2001a, 2001b) and for various smaller areas, including cattle stations and national parks (e.g. Albrecht & Pitts 2004; Bowman & Villiger 1995; Grant 1989; Latz 1995, 2007; Nelson 1985; Pitts & Matthews 2000), and some of these incorporate maps.

A significant characteristic of large parts of the study area is the extent and abundance of spinifex. Spinifex hummock grasslands occupy about a third of Australia and vast parts of the study area, and they are most prevalent in areas with infertile soils and low water holding capacity, for example, sands and gravels (Stafford Smith & Morton 1990). They occur in a wide variety of topographies and geologies, including sand sheets, sand dunes, and hilly areas of sandstone, quartzite or limestone. Individual spinifex plants have potential life-spans of decades but they are highly prone to fire due to their structure and resinous nature. Hummock grasslands may have a woody overstorey, but spinifex remains the characteristic species of this vegetation-habitat type because of its abundance, high coverage, proportion of overall biomass and influence on the presence of other plant and animal species.

Because spinifex-dominated vegetation is prone to relatively frequent and intense fire, associated species are typically either rapidly maturing or able to survive fire by resprouting or both (e.g. Duguid et al. 2009).

Spinifex is a strong competitor for moisture and nutrients (Nano & Clarke 2009), and consequently many species (annual, short-lived and perennial) rely on post-fire release from competition to grow and reproduce, resulting in a pulse in species richness following fire. Spinifex is often associated with mallee eucalypts, which resprout from a persistent lignotuber. Associations with fast growing *Acacia* shrubs are also common. Most of these fast growing acacias are obligate seeders that germinate following a fire and usually set seed before the next fire. In contrast, most of the taller trees growing with spinifex, such as bloodwood (*Corymbia opaca* (D.J.Carr & S.G.M.Carr) K.D.Hill & L.A.S.Johnson) and desert oak (*Allocasuarina decaisneana* (F.Muell.) L.A.S.Johnson), can resprout from branches.

Species that typically do not tolerate regimes of frequent intense fire rarely co-occur with spinifex. We define these as 'long-lived obligate seeders', which are slow to reach reproductive maturity (requiring one to two decades) and typically do not resprout when burnt. Such species can live for many decades, often more than a century, and are disadvantaged when fire recurs before a new generation has matured and set seed (e.g. Gill 2008, Griffin et al. 1983, Latz 2007). There are notable exceptions. Hill mulga (*Acacia macdonnellensis* Maconochie) is a long-lived obligate seeder that often grows over a spinifex understorey but typically occurs in areas with extensive out-cropping rock that can limit the spread of fire (Duguid et al. 2009).

Shrublands and low woodlands of obligate seeder species with little or no spinifex present are widespread across central arid Australia. *Acacia* woodlands are the most extensive of these, occupying about a quarter of arid Australia (Leigh & Noble 1969). They are typically dominated by mulga but also by various other structurally similar *Acacia* species (see Duguid et al. 2009) and grow with a diverse range of other woody shrubs and trees, for example, *Hakea* spp., *Grevillea* spp., *Senna* spp. and *Eucalyptus* spp. Many shrubland and woodland vegetation types rarely produce enough understorey fuel to carry a fire except in very wet years.

Mulga woodlands generally require 10–20 years without fire to become mature and re-establish their soil- and canopy-held seed banks. Much longer periods are required for mulga to grow tall enough to form woodlands and there is good evidence that older mulga stands provide habitat for a distinctive suite of birds (Leavesley 2008, Reid et al. 1993). Fire-free intervals of less than 15 years can result in a decline in the mulga component of the vegetation, and an increase in the more flammable grass species such as wiregrass (*Aristida* spp.) and other short-lived species (Latz 1990). When fires are of sufficient intensity to scorch the canopy of mulga woodlands, individuals are normally killed, although basal sprouting may occur in some limited circumstances (e.g. Duguid 2009). Mulga regeneration from seed appears to be enhanced following fire, especially if good follow-up rains occur. Regeneration can occur in the absence of fire (Williams 2001) but is typically less prolific than after fire.

Desert heath myrtle (*Aluta maisonneuvei* (F.Muell.) Rye & Trudgen) is another structurally dominant long-lived obligate seeder. It is a low to medium height shrub and grows on sandy soils. Unlike mulga, it rarely occupies extensive areas, tending to grow in relatively fine-scale mosaics with spinifex.

Other vegetation-habitat types occupying smaller areas of the study area include various types of grassland, shrubland, woodlands, stony plains, occasional herbaceous swamps and predominantly bare salt lakes and claypans. Mitchell grass plains dominated by *Astrebla* spp. occur in the semi-arid Barkly Tablelands to the north-east of our study area but are much more restricted within the arid area we have defined as central arid Australia. Where they do occur, Mitchell grasslands are highly fire prone, although grazing by domestic stock generally suppresses fuel loads. Some vegetation types rarely or never carry fire, including some chenopod shrublands, broad sandy river beds, claypans and salt lakes. Even so, these areas are significant influences on landscape-scale fire regimes, acting as natural fuel breaks which may halt the spread of a fire.

There are a number of plant species in central arid Australia that do not appear to occupy their potential range or are in very low abundance, possibly due to unfavourable fire regimes. For instance, white cypress pine (*Callitris glaucophylla* Joy Thomps. & L.A.S.Johnson) is a long-lived woody obligate seeder and individuals are susceptible to fire (e.g. Bowman & Latz 1993). Although cypress pine commonly co-occurs in areas dominated by spinifex, large trees almost always occur in small patches that are somewhat protected from fire by outcropping bare rock, and extensive woodlands of cypress pine are rare. However, it is not clear whether there is an ongoing decline in this species due to changing fire regimes or whether a long-term steady-state exists (Lee 2009).

Many native grass species other than spinifex and Mitchell grasses occur throughout the landscape and can contribute strongly to fuel loads. Perennials such as *Themeda* spp. and *Eulalia aurea* (Bory) Kunth can form dense understoreys in good conditions, particularly in run-on areas and riparian zones. Other native perennials (e.g. *Enteropogon* spp. and *Digitaria* spp.) can dominate the ground-layer in the more fertile areas. Following good rains, annual and short-lived native grasses (e.g. *Aristida* spp. and *Enneapogon* spp.) can become abundant and enhance fuel loads and continuity. Forbs and sub-shrubs can also add to fuel loads. In the absence of fire, some species are grazed by macropods, domestic stock and feral animals, while in areas free from domestic stock, termites are believed to be the major consumers of non-spinifex fuels (Latz 2007). Livestock grazing on pastoral lands limits fuel loads in all but the wettest, most productive years (Griffin & Friedel 1985, Latz & Griffin 1976).

Table 2: Common vegetation-habitat types in central arid Australia and key attributes influencing fire regimes

Notes: this information drawn from literature and expert opinion at the workshop. VL = very low, L = low, M = moderate, H = high, VH= very high; SG = spinifex grasses, PG = perennial grasses (non-spinifex, moderately long-lived), MG = mixed grasses (non-spinifex, mixed perennial and annual/short-lived perennial), AG = annual grasses (includes short-lived perennial), WCL = woody canopy and litter (leaves and fine twigs of trees and shrubs plus litter of fallen leaves), L = leaf litter, n/a = not applicable. 'Main fuel type' refers to fuel type following high rainfall periods – secondary fuels in brackets. '?' indicates uncertainty. Plant species nomenclature follows Albrecht et al. (2007).

Type	**Description**	**Proportion of regional area**[a]	**Soil nutrient availability**	**Main fuel type**	**Average fire frequency**	**Pastoral value**	**Potential for buffel grass invasion**	**Potential for ecological drift promoted by spinifex**	**Potential for termite fuel removal**
	DUNEFIELDS (dunes can be sparse and of low relief)								
1. Spinifex dunes and swales	Spinifex dunefields with scattered low trees and shrubs	VH	L	SG, (AG)	H	VL	VL	n/a	L
2. Desert oak woodlands	Open desert oak woodlands (*Allocasuarina decaisneana*) typically with a spinifex understorey (can be sparse or absent, particularly under the canopies of large trees)	H	L	SG, (L, AG)	M–H	VL	L–M	n/a	L
3. Canegrass-dominated dunes	Sand dunes dominated by canegrass (*Zygochloa paradoxa*), spinifex absent or sparse	L	L	PG, (?AG)	M	VL	VL	May go to type 1	?VL
4. Desert heath myrtle–dominated dunes	Typically dominated by desert heath myrtle (*Aluta maisonneuvei*) in dense stands of up to a metre or more tall	M	L	WCL	VL	VL	VL	May go to type 1	L
5. Non-spinifex, taller shrub dunes	Sand dunes dominated by drought-tolerant shrubs 1–3 m tall; typical species are witchetty bush (*Acacia kempeana*) and horse mulga (*Acacia ramulosa*)	L	L	WCL, (AG)	L–M	L	L	May go to type 1	M

Type	Description	Proportion of regional area[a]	Soil nutrient availability	Main fuel type	Average fire frequency	Pastoral value	Potential for buffel grass invasion	Potential for ecological drift promoted by spinifex	Potential for termite fuel removal
	PLAINS								
6. Spinifex plains with mixed overstorey	Spinifex with a variable overstorey, including mallee eucalypts and wattles; soils range from sandy to sandy clay loams	VH	L–M	SG, (AG)	H	L	VL	n/a	L–M
7. Mulga-dominated plains and dune swales with non-spinifex understorey	Plains dominated by mulga shrubland in various floristic and structural patterns; soils range from sandy loams to clay loams	H	(L–)M	WCL, MG	L(–M)	VL–M	VL–L	Some types may go to type 6 or 8	H
8. Mulga plains with spinifex understorey	Areas of mulga with spinifex understorey can form extensive patches (kilometres wide); typically in transition zones between non-spinifex mulga and spinifex sand plain	M	L–M	SG, (AG)	M	VL–L	VL	May go to type 6	H
9. Open grassy fertile woodlands	Characteristically occur on outwash plains surrounding hills and ranges; overstorey is frequently dominated by ironwood (*Acacia estrophiolata*), bloodwood (*Corymbia opaca*) and/or corkwood (*Hakea* spp.)	H	H	AG, MG	Variable, dependent on grazing	VH	VH	L	VH–H
10. Gidgee plains and dune swales	Gidgee shrubland dominated by georgina gidgee (*Acacia georginae*) or myall gidgee (*Acacia calcicola*) with intermediate textured, often calcareous soils	M	H–M	WCL, ?MG	L	M	M–H	May go to type 1 or 7	VH
11. Stony plains	Stony plains, principally gibber plains with very sparse vegetation cover consisting of mainly herbs, short grasses and scattered low shrubs	L	M–H	AG, MG	VL	M	L–M	L	L–M

Type	Description	Proportion of regional area[a]	Soil nutrient availability	Main fuel type	Average fire frequency	Pastoral value	Potential for buffel grass invasion	Potential for ecological drift promoted by spinifex	Potential for termite fuel removal
12. Chenopod plains	Plains dominated by chenopods, including saltbush (*Atriplex* spp.), bluebush (*Maireana* spp.) and burrs (*Sclerolaena* spp.); some species are strongly associated with saline and/or alkaline soils	L	M	WCL, AG	L, except where dominated by *Maireana aphylla*	M	M	L	L–M
13. Mitchell grass plains	Cracking clay soil plains dominated by Mitchell grass (*Astrebla* spp.).	L	M	MG	Variable, dependent on grazing	VH	L	L	M
	GRAVELLY RISES, HILLS AND RANGES								
14. Spinifex infertile non-calcareous hills	Areas of spinifex on quartzite, sandstone, laterite and silcrete	H	L	SG	H–VH	VL	VL	n/a	L
15. Non-spinifex infertile non-calcareous hills	Areas of non-spinifex shrubland on quartzite, sandstone and silcrete hills; dominants include *Acacia* spp. and cypress pine (*Callitris glaucophylla*)	M	L–M	WCL, MG	Variable, dependent on adjacent SG cover and patch size	VL	VL	May go to type 14	M
16. Spinifex fertile non-calcareous hills	Areas of spinifex on igneous or metamorphic geologies. Soils are typically shallow	M	M	SG	Variable, dependent on SG cover	L	L–M	n/a	L–M
17. Non-spinifex fertile non-calcareous hills	Shrubland on igneous or metamorphic geologies, dominated by *Acacia* spp., native fuchsias (*Eremophila* spp.) and *Senna* spp.	L	M–H	WCL, MG	H–VH	L–M	M–H	May go to type 16	M–H
18. Spinifex calcareous hills	Areas of spinifex (particularly *Triodia longiceps*) on calcareous hills	L	M	SG	L–M	L	?L	n/a	L

Type	Description	Proportion of regional area[a]	Soil nutrient availability	Main fuel type	Average fire frequency	Pastoral value	Potential for buffel grass invasion	Potential for ecological drift promoted by spinifex	Potential for termite fuel removal
19. Non-spinifex calcareous hills	Shrubland on calcareous hills, dominants include *Acacia* spp., native fuchsias and *Senna* spp.	VL	M	WCL, MG	??	M	H	May go to type 18	M–H
20. Breakaways without spinifex	Typically low shrubland on actively eroding hillslopes, often with saline soils	VL	L–M	WCL, AG	L	L–M	L–M	May go to type 14	M
	WETLANDS INCLUDING WATERCOURSES								
21. Riparian woodlands/ shrublands	River channels usually lined with river red gums (*Eucalyptus camaldulensis*) with a grassy understorey	L	H–VH	MG, L	M–H	VH	H–VH	L	M–H
22. Wooded swamps	Swamps with an overstorey of coolabah (*Eucalyptus coolabah* and *Eucalyptus victrix*) and variable understorey	L	M–H	L, MG	M–H	VH	L–M	May go to type 6	M–H
23. Non-wooded wetlands	Swamps and claypans lacking a tree layer but often with the shrubs lignum (*Muehlenbeckia florulenta*) or bluebush (*Chenopodium auricomum*); often herb-rich	VL	H	MG	Variable, dependent on surrounding vegetation	H	L–M	May go to type 6	L
24. Non-spinifex salt lake margins	Vegetated margins of salt lakes, often with samphires (*Tecticornia* spp.) or *Melaleuca* spp.	VL	L–M	WCL, MG	Variable, dependent on surrounding vegetation	M	L–M	May go to type 6	L–M

[a] Not all vegetation-habitat types are accurately mapped and many intermingle in a fine-scale mosaic (see Section 3), so that areal estimates are not readily quantifiable

4. Exotic grasses

By far the most important introduced plant species as far as fires are concerned is buffel grass. Buffel grass is native to Africa, the Middle East, Canary Islands, Madagascar, Indonesia, northern India and Pakistan (Tu 2000) and was introduced to arid Australia by cameleers in the 1870s (Friedel et al. 2006). Later, many varieties were imported in the anticipation that these would greatly increase pastoral productivity, and hence economic returns, and the grass is now spreading naturally across much of northern arid and semi-arid Australia (Friedel et al. 2006).

Buffel grass is a long-lived perennial (15–20 years), exhibiting high growth rates, early maturation, prolonged flowering periods, prolific seed production and highly effective seed dispersal mechanisms (Lawson et al. 2004, Miller 2003). Buffel grass is able to respond to both winter and summer rainfall (Clarke et al. 2005), which gives it a competitive advantage over native grass species. Buffel grass initially colonises areas with above-average soil moisture and nutrients, including flood-outs, channel areas, and run-on areas (Miller 2003). For example, in run-on areas, fuel loads (fine fuels < 6 mm diameter) averaged 1.8 t ha^{-1} in the absence of buffel grass, compared to 6.7 t ha^{-1} where buffel grass was present (Miller et al. 2010). In transitional areas containing buffel grass these fuel loads averaged 2–3 t ha^{-1}, whereas in run-off areas fuel loads in buffel-affected areas averaged 1–1.5 t ha^{-1} (Miller et al. 2010). From better-resourced areas buffel grass can spread naturally into surrounding areas (Friedel et al. 2006).

The invasive nature of buffel grass is widely recognised. The grass was introduced into Texas and northern Mexico in the 1930s and 1940s (Cox et al. 1988) and has since been documented in Arizona (where it has been declared a noxious weed), Hawaii and other USA states; in Sonora, Mexico; as well as in Puerto Rico, the Virgin Islands, South America and the West Indies, according to Tu (2000) citing numerous authors. It has also become dominant in regions of Africa where it is not native (D'Antonio & Vitousek 1992) and is spreading 'exponentially' in Mexico (Arriaga et al. 2004). Modelling based on rainfall, soils and elevation suggests the grass could cover up to 53% of the Mexican state of Sonora (Arriaga et al. 2004).

Lawson et al. (2004) assessed the potential distribution of buffel grass in Australia at a coarse continental scale, using a combined climate and soil suitability model. They estimated that 1.9 million km^2 (25%) of Australia was highly suitable for buffel grass, 3.3 million km^2 (43%) was moderately suitable, 1.1 million km^2 (15%) was marginal and 1.0 million km^2 (13%) was unsuitable. There were no data for the remaining 4%. Thus there is a significant possibility that buffel grass could colonise about two-thirds of mainland Australia.

Increased occurrence and intensity of fires in habitats colonised by buffel grass are widely reported (e.g. Arriaga et al. 2004, D'Antonio & Vitousek 1992, McDonald & McPherson 2011). Enhanced fuel continuity is likely following buffel grass colonisation in central arid Australia, conferring on susceptible vegetation-habitat types (Table 2) an associated risk of increased fire frequency and intensity. Overstorey species with fire-sensitive populations are thus vulnerable, particularly long-lived obligate seeder species, and consequently so is the habitat for dependent fauna. Buffel grass resprouts readily after fire and can rebuild fuel loads rapidly, especially if soils remain moist. It is possible that native plant species tolerant of short fire intervals will persist following buffel grass establishment but many species are thought to be disadvantaged due to competition and/or changed fire regimes. A similar situation is likely to occur with the invasion of couch grass (*Cynodon dactylon* (L.) Pers.) which occurs predominantly along watercourses.

5. Fire regimes in central arid Australia

Periodically, central arid Australia experiences widespread fire. For example, large fires occurred between 1918 and 1923 (Allan & Griffin 1986, Grant 1981, Griffin & Friedel 1985, Kimber 1983, Whitlock 1924), in the 1950s (Allan 1983), between 1974 and 1976 (Allan 1983, Allan & Southgate 2002, Luke & McArthur 1978), and between 2000 and 2003 (Allan 2009, Allan et al. 2003, Edwards et al. 2008, PK Latz unpublished data, JB Marsden-Smedley unpublished data, Russell-Smith et al. 2007, Turner et al. 2008). Several individual fires greater than 10 000 km^2 have been recorded over the past 35 years, including a 36 000 km^2 fire in the mid 1970s in the Simpson Desert (Allan 2009, Allan & Southgate 2002). In addition, between 1998 and 2004 almost 27% of arid and semi-arid Australia was burnt at least once (Turner et al. 2008). Fires initiated by lightning storms are common in the warmer months, and they can become large in the absence of fire management (Griffin et al. 1983). Large fires, although relatively infrequent, are particularly important in determining long-term average fire intervals due to their size (Gill & Allan 2008). There is potential for small fires in most years.

The northern parts of central arid Australia (e.g. northern Tanami and Davenport Murchison Ranges bioregions, Figure 1 and Table 1) are typified both by more frequent fires in general and more frequent occurrence of large fires, compared to the south. This corresponds with both higher rainfall and higher reliability of rainfall, which is also more strongly summer dominated. Allan and Southgate (2002) suggest fire-free periods average about five years for spinifex grasslands in the north of the study area and up to 20 years or more in the south, for example, the Great Victoria Desert (Figure 1 and Table 1).

The rate of recovery of spinifex grassland fuel loads correlates strongly with accumulated rainfall, rather than with time, and hence fire intervals can be as short as three years (Griffin 1984, Griffin et al. 1983, Wright 2007). These short-interval fires are generally limited to the areas burnt at the beginning of above-average rainfall periods and where subsequent rainfall contributes to rapid accumulation of mixed spinifex and non-spinifex fuel. In mulga woodlands there is usually only sufficient fuel to propagate a fire immediately following extended periods of high rainfall when there is a proliferation of short-lived and perennial grasses. Average inter-fire intervals are thus likely to be 20 years or more, although there are rare instances of fires within two years after years of high rainfall (Griffin et al. 1983). Where buffel grass thrives, both the fuel load and the fuel continuity increase and this has the potential to decrease fire interval and increase fire intensity (Clarke et al. 2005, Latz 1991).

Both Aboriginal and post-European settlement burning have been associated with managing the land for a range of objectives. Aboriginal people actively used fire for game management, access, communication, campsite clearing, ritual purposes and warfare, among other activities (Craig 1999; Giles 1889a, 1889b;

Gould 1971; Kimber 1983; Latz & Griffin 1976). Aboriginal fire management is believed by some observers to have produced a finer-scale mosaic of post-fire ages than occurs at present (Burrows et al. 2006) with the mean fire size possibly an order of magnitude smaller than that for the current fire regime (Latz & Griffin 1976), although Gill (2000) argues that the size of areas burnt was diverse, and Bird et al. (2008) observed that fine-scale mosaic burning was not uniformly distributed but scaled to foraging range and location of limiting resources such as water. After Europeans arrived, the movement of Aboriginal people to settlements and away from large portions of their country probably led to the development of extensive areas of long-unburnt spinifex grasslands. These areas now display an increased susceptibility to occasional large-scale, high-intensity fires (Burrows et al. 2006, Craig 1999, Gould 1971, Haydon et al. 2000).

6. Can optimal fire regimes be determined?

People attempting to manage vegetation–fire interactions may seek to define optimal or preferred fire regimes for particular vegetation types (Duguid et al. 2009). In theory, such regimes would allow particular species of plants and animals, and vegetation types, to persist. However, trade-offs are inevitable. The structure provided by older trees and shrubs is important for some animals, and the shelter provided by older spinifex hummocks is likely to be important for others. Conversely, recently burnt vegetation often has a higher abundance and diversity of some food plants. The elusive concept of optimal fire regimes must account for this diversity of ecological requirements while incorporating spatial factors (patch size) as well as temporal factors (time between fire) and the intensity of fire.

Any potential resolution of the question of optimal fire regimes was greatly hindered for us by the paucity of ecological data on plant species' life histories, seed biology, and habitat and food requirements of animals. Most critically, there were few quantitative data on time to reproductive maturity and factors that influence speed of maturation for plant species. Figure 3 shows indicative ranges of inter-fire interval that we estimated would be consistent with the perpetuation of some vegetation-habitat types. Intervals are those between fires of sufficient intensity to kill or damage overstorey species and trigger germination or resprouting. We recognised that fire regimes impact on species individually rather than on vegetation-habitat types as a whole, and so the ranges of values in Figure 3 are based on estimates of times to maturation of the species which define the vegetation-habitat types. The ranges of values allow for factors which may influence time to maturity, such as accumulated rainfall. Average fire frequencies (Table 2) were estimated from our knowledge of inter-fire intervals such as these. Thus an inter-fire interval of:

- 5–30 years for (1), sand dune spinifex grassland, equated to a high (H) average fire frequency
- 20– >50 years for (4), desert heath myrtle, was equivalent to a very low (VL) average fire frequency.

The vegetation-habitat types dominated by obligate seeder species are all thought to persist without fire for intervals exceeding a century. Infrequent but intense fires may be necessary for the persistence of the current vegetation type. In the absence of fire, floristic composition might shift gradually, which we indicate with a question mark for the optimal upper limit to the interval between fires in Figure 3. Some specific successional changes in vegetation composition and structure following an intense fire have been documented for central arid Australia (e.g. Clarke et al. 2005, Griffin 1984, Latz 1990, Rice & Westoby 1999). Latz (2007) has proposed that with very long periods without fire, the floristic composition of overstoreys may become more diverse. In an area initially dominated by mulga, for example, other species may become co-dominant due to accessions of bird-dispersed seed. However, there are few data to support the concept of predictable change towards a climax vegetation type, and optimal fire regimes remain elusive. In reality the interval between fires for a given patch of a vegetation-habitat type is influenced not only by rainfall history and growth characteristics of the dominant fuel, but also by grazing and the flammability of adjacent vegetation types.

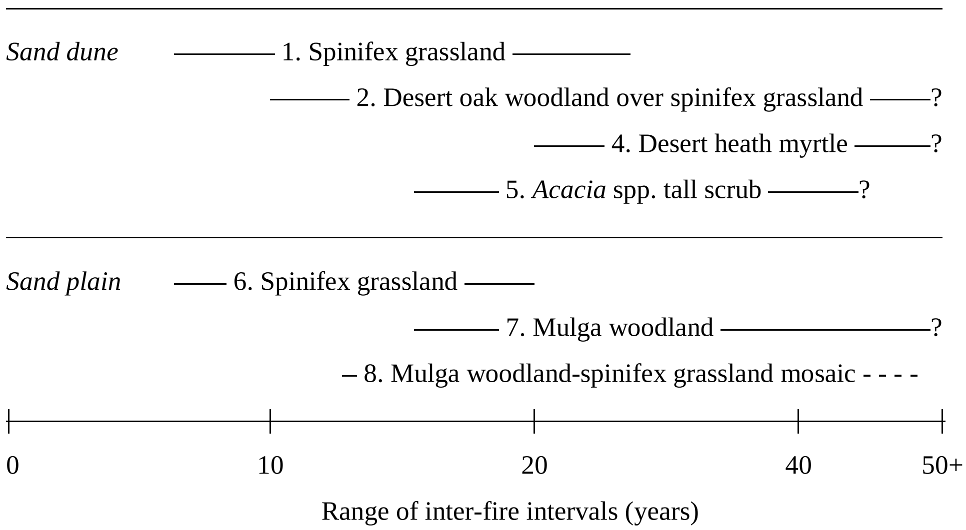

Figure 3: Proposed inter-fire intervals

Proposed range of inter-fire intervals for perpetuation of different vegetation-habitat types in sand dune and sand plain landforms, based on estimated time to reproductive maturity of defining species and factors such as accumulated rainfall,which may influence time to maturity. Numbers refer to vegetation-habitat types in Table 2.

7. Towards a conceptual framework for central arid Australian vegetation–fire interactions

An understanding of the main drivers of vegetation change of central arid Australia is critical for good land management, whether its objectives are to maintain biodiversity, ecosystem functioning, pastoral activities and/or Aboriginal values. Presently, a comprehensive understanding of vegetation–fire interactions is lacking for this area. Here we summarise and collate current vegetation–fire knowledge from arid Australia in a conceptual framework that enables the relationship between average fire frequency (or its inverse, average fire interval) and potential fuel loads in different vegetation types to be explored. This framework should encourage debate and stimulate further research, ultimately leading to better land management.

7.1. Vegetation–fire interactions

Figure 4 summarises our current understanding of vegetation–fire interactions. The vegetation-habitat types detailed in Table 2 are positioned in Figure 4 according to their relative nutrient and moisture status and their typical fire frequency. The potential fuel load and average fire frequency are greatest in spinifex grasslands (vegetation-habitat types 1, 2, 6, 8, 14, 16, 18), and least on gravel surfaces with minimal vegetation (11, 20). The fertile plains (9, 13, 21) can support high fuel loads after major rainfall events, but their grass fuels will be removed by termites and/or decay more rapidly than spinifex fuels. In addition, they lack the fire-enhancing resins of spinifex, so that fire will occur less often on the fertile plains. Fuel loads on non-spinifex woodlands and shrublands (3, 7, 10, 12, 17, 22, 23) and non-spinifex shrublands (4, 5, 15, 19, 24), are less than those on fertile plains and spinifex grasslands, and yet some woodland and shrubland vegetation-habitat types support relatively high average fire frequencies. Such high fire frequencies are due to the proximity of these vegetation-habitat types to extensive spinifex grasslands, rather than to their inherent flammability. For example, canegrass dunes (3) intermingle with spinifex-dominated interdune corridors; saline saltbush and samphire shrublands (24) usually form relatively narrow patches on salt lake margins, and are subject to the fire regimes of adjacent spinifex grasslands; and desert heath myrtle shrublands (4) often intermingle with spinifex grasslands.

Several vegetation-habitat types (9, 19, 21), are prone to invasion by buffel grass because they are well-supplied with nutrients and water. Expansion of buffel grass into new areas of these vegetation-habitat types is continuing, and hence they are likely to shift to higher potential fuel loads and higher fire frequencies. Some woodlands and shrublands (4, 5, 7, 15, 17), as well as mulga plains with spinifex (8), may be prone to encroachment by spinifex after fire. If this eventuates, higher and more flammable fuel loads will result and fire frequencies will increase.

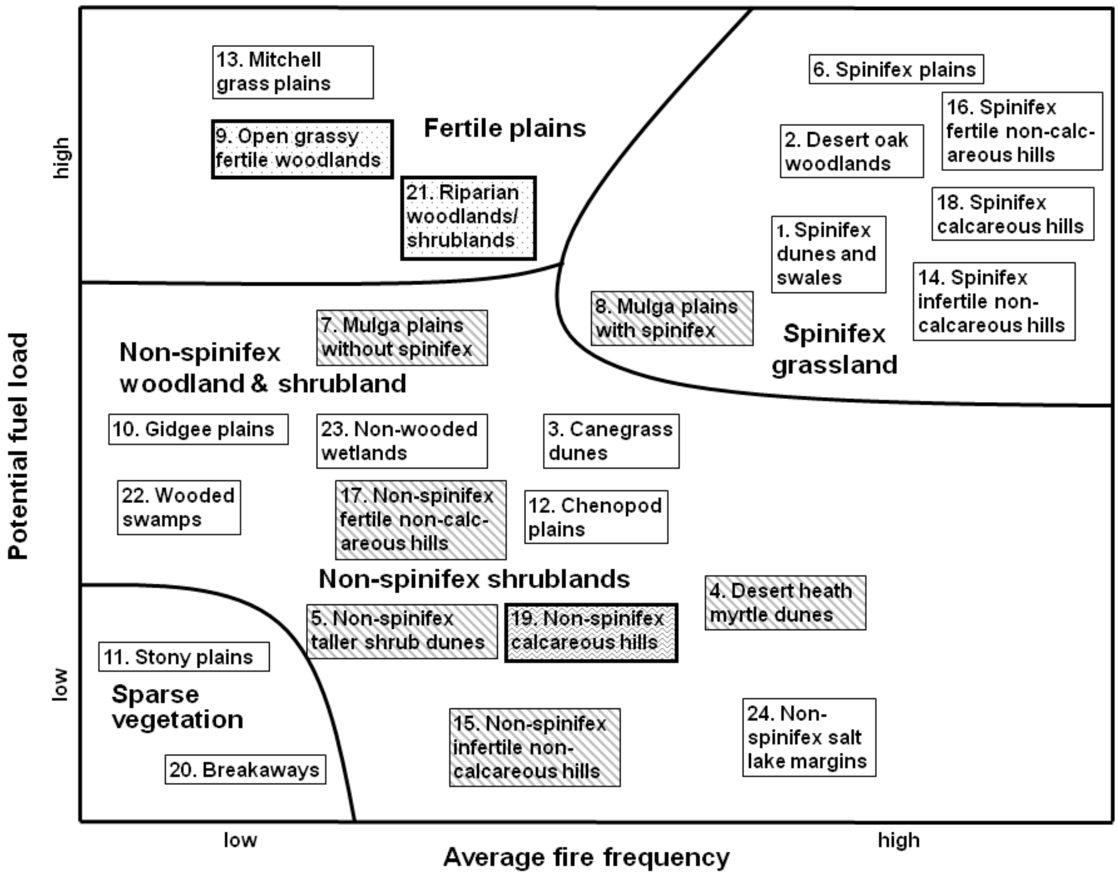

Figure 4: Vegetation-habitat types against axes of potential fuel load

The current relative positions of vegetation-habitat types detailed in Table 2 on axes of potential fuel load in above average rainfall seasons and average fire frequency. Bold outlined box and stipple = highly prone to buffel grass promoted change in vegetation-habitat type and fire frequency. Bold outlined box and ripple = highly prone to buffel grass promoted change, and may be prone to spinifex promoted change, in vegetation-habitat type and fire frequency. Hatching = may be prone to spinifex promoted change in vegetation-habitat type and fire frequency.

There are a number of examples from other landscapes of conceptual vegetation–fire interaction modelling which could assist in the further development of a conceptual vegetation–fire framework for arid Australia. For example, in South African C_4 grassland ecosystems, Keeley and Rundel (2005) proposed a disturbance-frequency model which suggested that high disturbance frequencies are required for grasses to out-compete trees. They postulated that in arid woodland areas, only moderate levels of disturbance (e.g. fire) are required to shift the vegetation to grassland. Further, Bond and Keeley (2005) described the consumption of biomass by fire as analogous to consumption by herbivores. They presented a model that attempted to explain the divergence between the vegetation expected at a site according to its climatic and edaphic potential versus the vegetation that actually occurs, with this divergence being an indication of the relative importance of fire versus other vegetation-consuming factors affecting the site. In an Australian context, Jackson (1968, 1978) proposed a conceptual model for understanding vegetation patterns of the wetter parts of Tasmania. He suggested that the pattern of plant communities there was a response to limitations of soil fertility and frequency of fires, and that climate influenced the vegetation indirectly by the interactions of fertility and fire frequency. Infertile soils were strongly correlated with frequent fire due to loss of nutrients through fire as well as the development of fire-prone communities. Jackson (1978) proposed that the distribution of fires led to a process of 'ecological drift', in which 'any given area moves gradually towards climax rainforest or towards sedgeland'.

7.2. Potential 'ecological drift' and climate change

Similar shifts in vegetation have been postulated for central Australia. Cleland (1966) and Latz (2007) suggested that spinifex may be invading mulga shrublands as a consequence of an increase in incidence of fire, whereas Chippendale (1958) suggested that mulga may be invading spinifex grasslands. In central Australia, mulga woodlands and spinifex grasslands are generally well-delineated, and both occur as very large patches of relatively homogeneous vegetation, although more fine-scale mosaics of the two also occur. There is some evidence that mulga–spinifex boundaries fluctuate over a period of decades as a consequence of fire and recovery from fire. Several observers, including some authors of this paper, suggest that these shifts can persist in the longer term but this requires quantitative evidence. Existing studies of mulga–spinifex boundaries (Bowman et al. 1994, 2007, 2008; Nano & Clarke 2008; Nicholas 2007) have found little evidence of contraction of mulga over longer time frames, but whether these studies have general applicability is unknown.

A long-term increase in rainfall could lead to a more rapid accumulation of spinifex fuel and increased fire frequency, preventing recovery of mulga woodlands at their margins and causing their contraction. However, Wright and Clarke (2007) concluded that short fire intervals in hummock grasslands were not likely to cause major declines in dominant acacias, including mulga.

Species such as desert heath myrtle can grow in relatively fine-scale mosaics with spinifex, and hill mulga grows over a dense spinifex understorey. Some observers have expressed concern that populations of these species are being diminished by current fire regimes, but quantitative data are few and inconclusive. It would be reasonable to consider that these species might also be disadvantaged by a long-term increase in fire frequency.

There are other vegetation-habitat types such as hills with cypress pine, which may already be 'drifting' towards a long-term loss of the pine overstorey due to the lack of traditional Aboriginal burning and the resultant increase in high-intensity fires. However, the only study to examine this (Bowman & Latz 1993) was based on a single survey at a point in time, and the authors noted that assessment over decades was required to confirm that the observed contraction of cypress pine distribution was a long-term feature. In the event that it is, and the high susceptibility of cypress pine populations to fire suggests that it could be, increased rainfall could exacerbate the situation.

We have observed that older unburnt patches in spinifex grassland tend to have both denser and taller tree and shrub layers, often dominated by fast growing *Acacia* species and resprouting species of *Eucalyptus* and *Corymbia*. We have also observed spinifex dying out under long unburnt mulga. Hence the classification of an area's vegetation as woodland, shrubland or spinifex grassland may be a matter of time since fire. Beyond that, it is possible to speculate that, in the complete absence of fire or drought, spinifex may eventually diminish, to be replaced by a 'climax' community of woodland or shrubland in the sense of Jackson's (1978) proposed 'ecological drift', but such a proposition is largely untestable, due to the inevitability of fire within the time required for community replacement.

If changes in fire frequency occur, for example through climate change, then vegetation associations could shift but under different drivers compared with the 'ecological drift' model proposed by Jackson (1978). We have discussed the potential impacts of increased rainfall. On the other hand, recent climate change forecasts (CSIRO 2007) indicate declining rainfall for the study area is possible, and McKeon et al. (2009) suggest that any beneficial effects of increased CO_2 on plant production would likely be negated by forecast increases in temperature, so that increased fire frequency would seem unlikely. Modelling climate change effects on fire regimes for a central arid Australian study site by one of us (KJ King unpublished data) shows that with warmer and drier future climates in arid Australia, fire incidence and areas burned are likely to decrease, and there is a lower probability of very large fires.

7.3. Potential buffel grass invasion, tree/shrub balance and climate change

The better-resourced areas (Figure 4: vegetation-habitat types 9, 19, 21) are particularly susceptible to invasion by buffel grass if rainfall and fire frequency increase (Table 2). Should there be an increase in rainfall, or in the frequency of high rainfall events, greater accumulation of buffel grass fuel is likely to promote more frequent, often higher intensity fires and consequent loss of overstorey trees and shrubs (Miller et al. 2010). Alluvial plains with open grassy woodlands will be particularly vulnerable, due to the extra run-off they receive from nearby hills and ranges. Margins of waterways are already at risk and will continue to be under threat from higher intensity and more frequent fires resulting from buffel grass invasion (INRMP-NT 2005, NT-NRETAS 2007). As noted earlier, old hollow-bearing river red gums are particularly threatened, with inferred consequences for dependent fauna. Hybridisation and local adaptation of buffel grass may be occurring already in central Australia (Friedel et al. 2006), which suggests that other less favourable vegetation-habitat types such as quartzite hills may also be colonised over time. How the interaction of warmer, drier conditions and increased CO_2 might play out for buffel grass, a C_4 species, under climate change is unknown.

In the semi-arid grasslands of Colorado, which support both C_3 and C_4 grasses, doubling CO_2 above ambient increased biomass production as a consequence of better water use efficiency, but only in one of the C_3 grasses (Morgan et al. 2004). Since buffel grass is a C_4 species, elevated CO_2 may not enhance its productivity. Morgan et al. (2004) also found that digestibility decreased with elevated CO_2 in all the grasses they studied, and hence potentially lowered forage quality and animal production. Shrubs, on the other hand, were favoured (Morgan et al. 2007) and could lead to an expansion of woody plant populations. Berry and Roderick (2004) also predicted an increase in water use efficiency in Australian evergreen communities (including trees and shrubs) with increasing CO_2, and SL Berry (personal communication to AM Gill 2008) considered it likely that tree canopies would carry more leaf material. This suggests that there will be more litter fall if leaf longevity does not increase. Hovenden and Williams (2010) predicted that the nitrogen content of the litter will decrease so we may expect the decomposition rate to slow. Thus, there is potential for the litter fuel load to increase, promoting fuel continuity and the potential for fire spread across landscapes. There may be an increased role for fire in reducing shrub biomass and maintaining grass pastures for livestock, but the dynamic balance between grass and woody plants, under different scenarios of decreased rainfall and increased CO_2 in an already highly variable climate, warrants further investigation.

Increasing density and dominance of trees and shrubs has been noted in arid and semi-arid ecosystems worldwide (Archer et al. 1995) and observers locally recognise 'woody thickening' in some vegetation-habitat types of central arid Australia. While it may be tempting to attribute woody thickening to increasing CO_2 and rainfall, causes are not clear. Archer et al. (1995) argued generally that, while changes in these factors may have occurred at a regional scale, they appeared to be insufficient to produce the vegetation changes that occurred at smaller scales. While changes probably involved interactions between climate, atmospheric CO_2 enrichment, fire and grazing, evidence generally indicated that livestock grazing was the proximate cause through its direct effects and its indirect effects, which included fuel reduction and consequent decreases in fire frequency and/or intensity.

An in-depth discussion of the role of grazing livestock, native and feral vertebrates and termites in altering fuel accumulation is beyond the scope of this paper. However, it would be reasonable to propose that the greatest impact of non-native grazers will continue to be in the more nutrient rich vegetation-habitat types, regardless of rainfall projections. Testing of this proposition will be confounded by spatial and temporal patterns of rainfall and differences among fuel types.

7.4. Conclusions

This paper has integrated published literature and expert knowledge to produce an initial conceptual framework for vegetation–fire interactions in central arid Australia and has discussed the ways in which these interactions may change as a consequence of climate change and the spread of an introduced grass. There are many gaps in our current knowledge. Our purpose has been to provide a basis for further thinking and empirical studies designed to improve the current understanding of vegetation–fire interactions in central arid Australia.

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