

Drivers, ecology, and management of fire in fynbos

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3.1 Introduction

Fire is a fundamental driving force in fynbos dynamics, being both inevitable and necessary in fynbos ecosystems (Kruger and Bigalke 1984). Fire has also played a key role in stimulating the evolutionary diversification of the Cape flora (Cowling 1987; Keeley et al. 2012), with contemporary fire regimes probably dating back to the Middle Miocene climatic optimum (c.15 Ma; Bytebier et al. 2011). In modern fynbos, fire is the dominant natural disturbance, and managing fynbos equates to managing fire. This chapter outlines the determinants of fire and fire regimes in the Cape Floristic Region (CFR), the biotic responses and adaptations it elicits, and fire management.

3.2 Biophysical determinants of fire

For any fire to start there must be (a) sufficient fuel in a continuous fuel bed to allow fires to spread; (b) warm, dry weather; and (c) a source of ignition. The absence of ignition sources can limit the number of fires occurring (Seydack et al. 2007; Archibald et al. 2009), while weather conditions influence fire behaviour and the area burnt (Moritz 2003; van Wilgen et al. 2010). Only a small fraction of lightning ground strikes result in ignition of vegetation. Lightning density is therefore not a good indicator of fire frequency (Keeley et al. 1999), although the temporal distribution of lightning may provide an indication of the seasonality of natural fires. Lightning is not the only ignition source, with humans having burnt fynbos since at least the start of the Holocene 12 000 years ago to maintain productive stocks of geophytes and to facilitate hunting (Deacon 1983).

3.2.1 Fynbos vegetation as fuel

Properties rendering plants more or less susceptible to fire are the production or retention of dead material; the shape, size, and arrangement of plant parts (determining rate of moisture loss and the ratio of fuel to air); and their chemical composition (energy and volatile compound content). In mature fynbos, ericoid and proteoid shrubs, herbaceous plants, and standing dead material form a vertically continuous fuel bed about 1–2 m deep, commonly supporting fires in the crowns of dominant shrubs (van Wilgen, Higgins, et al. 1990). The finer restioid and ericoid fuels make fynbos susceptible to burning at any time of the year and seasonal curing of herbaceous vegetation does not affect fire seasonality in fynbos as it does in grasslands and chaparral (van Wilgen 1984a).

Fynbos is less flammable than Californian chaparral or Australian *Eucalyptus* woodlands due to lower crude fat contents (c.0.4–10.3%) and higher foliar moisture contents (c.98–126%) that do not appear to vary widely between seasons (van Wilgen, Higgins, et al. 1990). Net primary production in fynbos is 1–4 tonnes ha⁻¹ yr⁻¹ and is typically highest during the first three years post fire (Kruger 1977). Total above-ground phytomass increases with post-fire age, from 6–9 tonnes ha⁻¹ 4 yr post fire, to 6–15 tonnes ha⁻¹ 10–19 yr post fire, and 11–76 tonnes ha⁻¹ ≥20 yr post fire (Stock and Allsopp 1992). Growth rates in fynbos are affected by soil moisture availability more than by soil fertility (Kruger 1977). In fynbos of <5 yr post-fire age, all the above-ground phytomass constitutes potential fuel (i.e. shoots <6 mm diameter), whereas at least half the above-ground phytomass comprises potential fuel in >20 yr old fynbos (van Wilgen and

van Hensbergen 1992). Annual litter production in fynbos is $c.0.8$ tonnes ha^{-1} in western coastal fynbos (Mitchell et al. 1986). The litter layer was found to increase with post-fire vegetation age due to extremely slow decomposition rates (predicted turnover times of 11–15 yr; Mitchell et al. 1986), suggesting that fire, rather than decomposition, is the major means of nutrient release (Stock and Allsopp 1992). However, it was recently shown that fynbos and renosterveld taxa exhibit a wide range in decomposition rates (8-fold and 20-fold differences, respectively, amongst plant functional types), suggesting that biological decomposition may be important for carbon and nutrient cycling between fires, particularly in the initial post-fire years (Bengtsson et al. 2012).

3.2.2 Fire climates and weather

Temperature and relative humidity strongly influence fuel moisture, which in turn determines whether and how the vegetation will burn, while wind velocity drives the spread of fires. Fire danger rating systems (e.g. Noble et al. 1980) track how these weather variables affect the likelihood of fires starting and spreading and are used to characterize fire climates based on fire potential during the fire season (van Wilgen 1984a; van Wilgen et al. 2010; Kraaij, Cowling, and van Wilgen 2013a). Other approaches use meteorological sequences depicted by synoptic states (Juhnke and Fuggle 1987; Southey 2009). Van Wilgen (1984a) discerned five fire climate zones (i.e. western, southwestern, and southeastern coastal zones, and western and eastern inland zones; Fig 3.1) for the CFR. Southey (2009) confirmed this delineation, while Kraaij, Cowling, and van Wilgen (2013a) differentiated a sixth (far eastern coastal) zone. The severity of fire weather distinguishes inland fire climate regions from coastal regions, where weather conditions are milder (Figs 3.1 and 3.2, van Wilgen et al. 2010). The west–east division is based on the seasonality of fire weather, with a unimodal cycle of high fire danger in summer in the west and bimodal or aseasonal patterns in the east (Fig 3.1, Southey 2009; Kraaij, Cowling, and van Wilgen 2013a). Fires in the west are associated with easterly wave low pressure systems, characterized by strong southwesterly winds, convective activity and lightning, while in the east, fires are often associated with hot, dry, katabatic winds ('berg winds') that precede tropical temperate troughs (Southey 2009). In the western and southwestern coastal zones, maritime amelioration makes the seasonal difference less marked (van Wilgen 1984a). In the eastern coastal zones the trend is reversed with fire

potential being highest in winter (associated with berg winds), although there is little seasonality, and low or moderate fire danger conditions are the norm, particularly in the far east (Kraaij, Cowling, and van Wilgen 2013a). The eastern inland zone also experiences rainfall throughout the year, but high evapotranspiration in summer results in a predominantly summer fire regime. The western inland zone exhibits the strongest seasonal trends, with high fire potential being virtually restricted to summer (van Wilgen 1984a).

Although fynbos fires take place under a wide range of weather conditions, fire size is positively correlated with fire danger index (FDI) on the date of ignition or the highest FDI recorded during the course of the fire (van Wilgen et al. 2010; Kraaij, Cowling, and van Wilgen 2013a). Fires of >5000 ha in the western and inland part of the CFR are always associated with FDI >10 (moderate to high) and fires >10 000 ha associated with FDI >20 (high to very high; van Wilgen et al. 2010), whereas in the southeastern coastal CFR, weather conditions associated with the largest fires were variable rather than consistently extreme (Kraaij, Cowling, and van Wilgen 2013a). In the east, FDIs associated with fires in spring and summer were lower than those in autumn and winter (Kraaij, Cowling, and van Wilgen 2013a; Kraaij, Baard, et al. 2013).

3.2.3 Lightning and other ignition sources

Despite relatively infrequent thunderstorms and the low density of lightning in the CFR ($0.2\text{--}3.4$ km^{-2} yr^{-1} ; study period 1975–1981) compared to that in grassland and savanna (Manry and Knight 1986), lightning is a common cause of fires (Seydack et al. 2007; van Wilgen et al. 2010; Kraaij, Baard, et al. 2013). Little is known about geographic and temporal variation in lightning incidence in the CFR (Keeley et al. 2012). A first detailed study of lightning occurrence (based on a continuous detection network, 2006–2010) in fynbos revealed that lightning density was significantly higher in the eastern coastal zone (0.42 ± 0.09 strikes km^{-2} yr^{-1} , mean \pm 95% confidence intervals) than in the western coastal zone (0.16 ± 0.03 strikes km^{-2} yr^{-1} ; Kraaij, Cowling, and van Wilgen 2013a) and appeared to increase towards the interior (Fig 3.3a). Lightning seasonality did not differ markedly along an east–west gradient in the CFR, occurring throughout the year across the CFR with somewhat reduced activity during the winter months (June–September; Fig 3.3b, Midgley 1989; Kraaij, Cowling, and van Wilgen 2013a). Midgley (1989) correlated the paucity of natural fires during winter in the Swartberg with diminished lightning

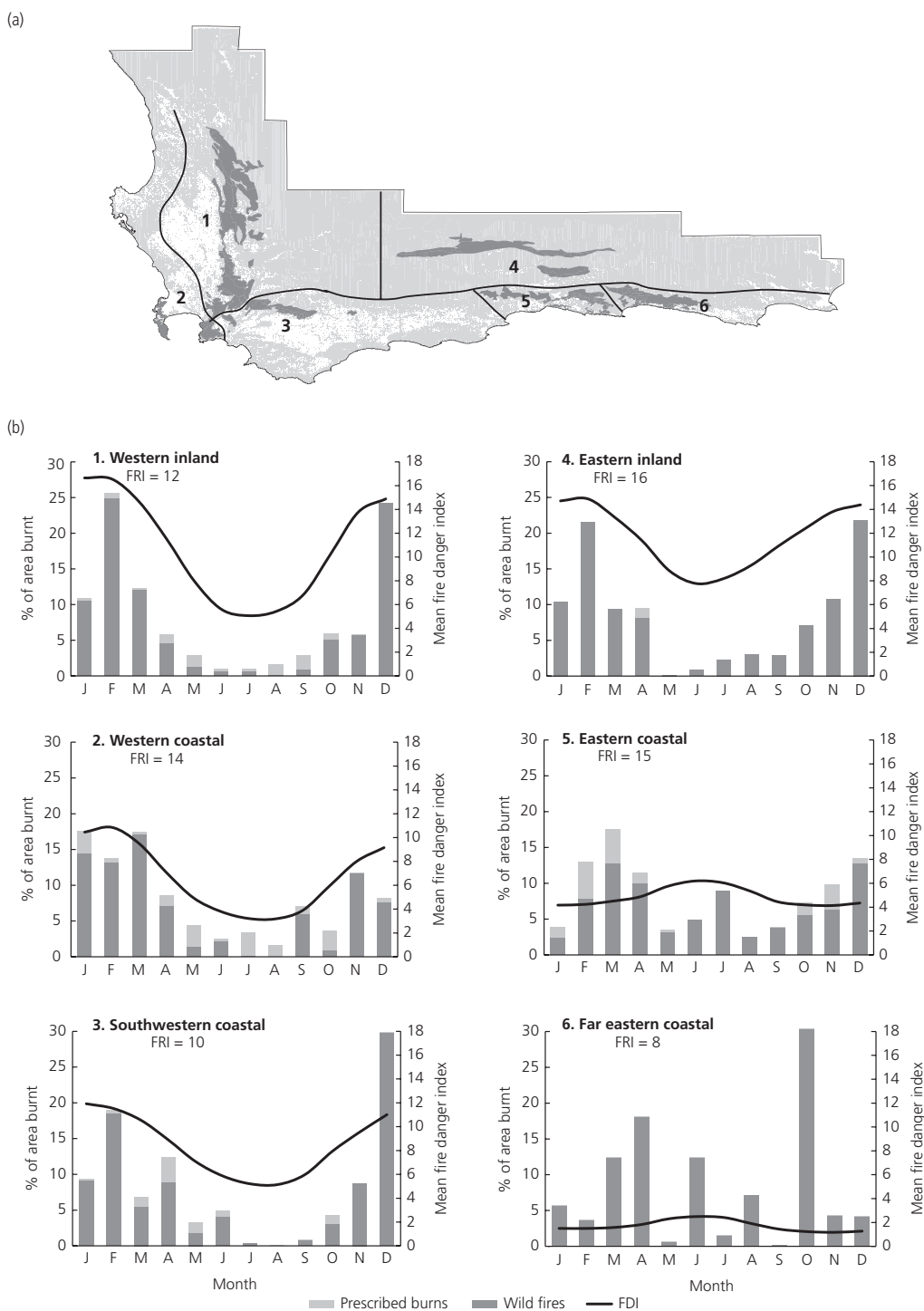


Figure 3.1 (a) Fire climate zones in the Cape Floristic Region (adapted from van Wilgen 1984a) also showing the extent of untransformed vegetation (light grey) and protected areas (dark grey). (b) Monthly distribution of area burnt in wildfires (dark grey bars) and prescribed burns (light grey bars) in protected areas (grouping and timespan follow Table 3.1) per zone in relation to mean monthly McArthur’s Forest Fire Danger Index (solid lines, after van Wilgen et al. 2010; Kraaij, Baard, et al. 2013; Kraaij, Cowling, and van Wilgen 2013a; timespan follows Fig 3.2). FRI = the mean fire return interval for each climate zone.

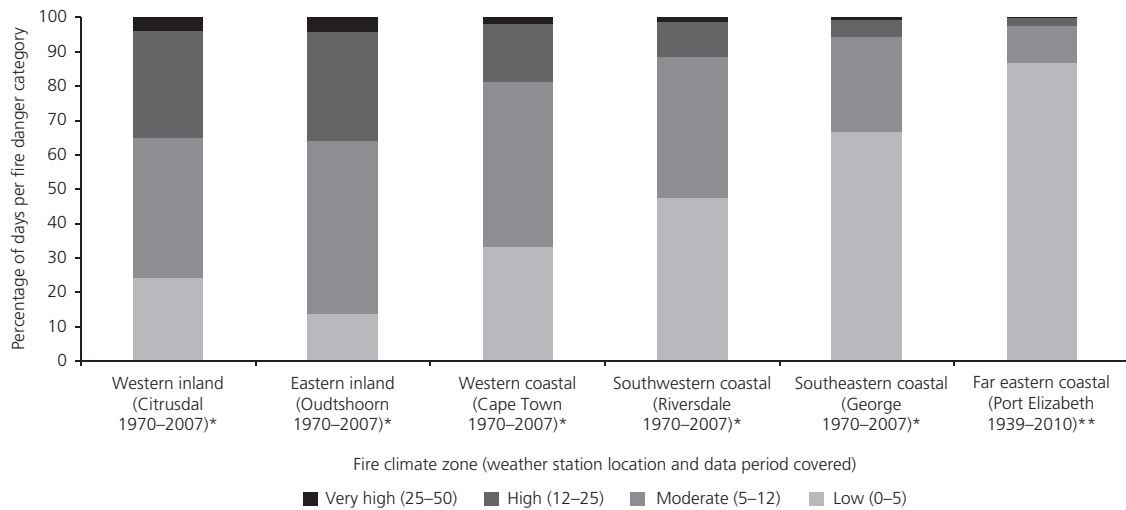


Figure 3.2 The percentage of days within different fire danger categories in six fire climate zones in the CFR. Fire danger indices and ratings are from the McArthur Forest Fire Danger Rating system (Noble et al. 1980). Data from *van Wilgen et al. 2010 and **Kraaij, Cowling, and van Wilgen 2013a.

activity during this time, although the lower frequency of fire in winter is more likely due to high fuel moisture (van Wilgen 1984a). Fire incidence and peak lightning seasons are not correlated in southern African biomes (Archibald et al. 2009), as much lightning occurs when conditions are too humid to support fire. In the eastern coastal CFR, seasonality in lightning fires was correlated with seasonality in lightning occurrence when assessed over five years, suggesting that a lack of lightning may well limit fire occurrence, contrary to the findings of Archibald et al. (2009). However, the correlation was not apparent in the long term (70 yr), suggesting that the fire season is variable in time in this region and not concentrated in the same calendar months every year (Kraaij, Baard, et al. 2013; Kraaij, Cowling, and van Wilgen 2013a).

Although lightning would have been the only source of ignition in the CFR for millennia, humans have provided a source of ignition for tens of thousands of years (Chapter 8, Deacon 1983). In some fynbos areas, lightning remains the dominant ignition source (Seydack et al. 2007; Kraaij, Cowling, and van Wilgen 2013a), but humans are the foremost cause of ignition in most places and this is set to grow as human populations grow. Human ignitions could change the seasonality and increase the frequency of fires, leading to concerns about the effects of these changes on the conservation of fynbos ecosystems (Forsyth and van Wilgen 2008; Southey 2009; van Wilgen et al. 2010).

3.2.4 Climate change and fires

Projections of global changes to climate suggest that much of the CFR will become hotter and drier (Chapter 13, Midgley et al. 2003; Hoffman et al. 2011), which has definite implications for fire regimes. Fire frequency is likely to increase under predicted changes in temperatures and rainfall (Wilson et al. 2010). Synoptic states associated with fire occurrence (and in particular, strong high pressure systems leading to berg winds) have increased in frequency during the past four decades in the CFR (Southey 2009). Mean FDI and the number of days with moderate or higher fire danger ratings have increased significantly since the 1940s in the southeastern CFR (Kraaij, Cowling, and van Wilgen 2013a), associated with decreases in minimum relative humidity and increases in temperature. Trends in rainfall and wind speed were more variable amongst weather stations. In an assessment (1974–2005) of 20 weather stations situated throughout the CFR, wind run declined at all stations, pan evaporation declined at 16 stations, maximum temperature increased at all but one station, minimum relative humidity decreased at half the stations, and annual rainfall did not change significantly (Hoffman et al. 2011). Contrary to predictions of increasing fire frequency and severity based on observed or predicted increases in temperature and windspeed, and decreasing rainfall and relative humidity (Wilson

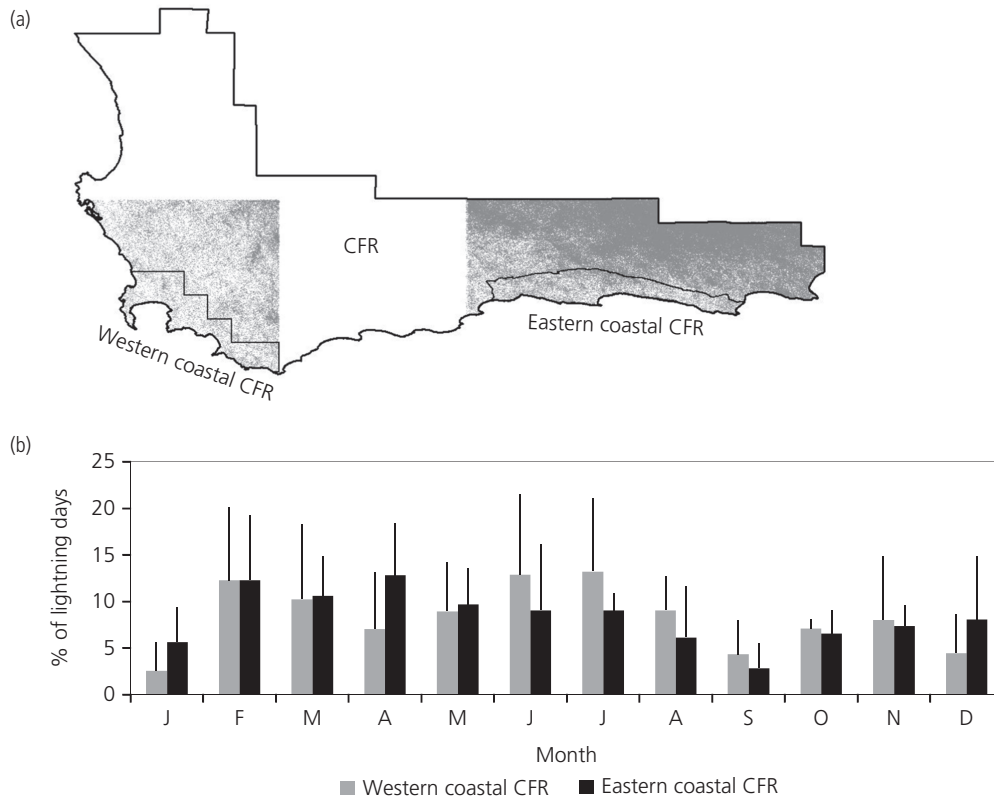


Figure 3.3 (a) Lightning ground strike distribution (2006–2010; each strike depicted by a dot) over the western and eastern coastal zones (boundaries used in analysis are outlined in black) of the CFR. (b) Seasonality of lightning expressed as the percentage of the annual number of lightning days occurring per month. Means \pm 95% confidence intervals across years are shown for the same two regions. (Adapted from Kraaij, Cowling, and van Wilgen 2013a; data courtesy of the South African Weather Service).

et al. 2010; Kraaij, Cowling, and van Wilgen 2013a), reductions in wind run and evaporative demand could reduce fire frequency and intensity (Hoffman et al. 2011). The implications of climate change are therefore not clear, but obviously a cause for concern. Inferring changes in fire regimes from changes in climate (as opposed to other direct human influences) is tricky and requires comprehensive historical data. Attempts are limited to the work by van Wilgen and Scott (2001), Wilson et al. (2010), and Kraaij, Baard, et al. (2013). Complex interactions between disturbance regimes, plant life history traits, and distribution patterns may also mediate whether particular species will be exposed to increased extinction risks under future climate scenarios, underscoring the need for methods that link spatial and demographic processes (Keith et al. 2008).

3.3 Recent fire regimes

The pattern of fire occurrence over an extended period in a given area is referred to as a fire regime, defined as the typical frequency, seasonality, intensity, and size of fires (Gill and Allan 2008). The reconstruction of past fire regimes can contribute to improved understanding of the ecological role of fire. Comprehensive accounts of fire histories in the CFR are confined to protected areas and have only recently been documented in the primary literature (Brown et al. 1991; Seydack et al. 2007; Forsyth and van Wilgen 2008; Kraaij 2010; van Wilgen et al. 2010; Kraaij, Baard, et al. 2013), facilitated by the development of geographic information systems. These accounts have enabled characterization of fire regimes instead of single fire events, and assessments of the influence of fire management policies and

interventions. Satellite-derived burnt-area products hold promise for wide-scale, affordable future mapping of fire scars (De Klerk et al. 2012).

3.3.1 Fire frequency

Estimates of average fire return intervals (FRIs) in CFR protected areas during the past 30–80 years range from 7–55 yr (mostly 10–20 yr), with shorter FRIs recorded in the Tsitsikamma region and lowland fynbos of the Bontebok National Park, and longer FRIs in drier areas of the Swartberg Mountains (Table 3.1, Fig 3.1). Although a useful measure, averages of FRI obscure the variability in burning patterns. This variability can have important ecological effects. Local or regional moisture regimes affect plant growth rates, and fires generally become more frequent along rainfall gradients (e.g. from west to east in the eastern coastal region (Kraaij, Baard, et al. 2013) and along an altitudinal rainfall gradient in the Swartberg (Seydack et al. 2007)). In the latter, FRIs in dry shrublands at lower elevations were controlled by the rate of fuel accumulation, whereas lightning occurrence and weather constituted the dominant controls at higher elevations (Seydack et al. 2007).

Globally and in the CFR, many areas have experienced recent increases in fire frequency or in the occurrence of large fires (Keeley et al. 1999; Forsyth and van Wilgen 2007, 2008; Seydack et al. 2007; Kraaij, Baard, et al. 2013). These have been associated with increases in the frequency of weather conditions favourable for fires (van Wilgen and Scott 2001; Mouillot et al. 2002; Wilson et al. 2010; Kraaij, Cowling, and van Wilgen 2013a), or increases in human densities and related ignition sources (Keeley et al. 1999; Forsyth and van Wilgen 2008; Southey 2009; van Wilgen et al. 2012), causative factors being hard to discern.

The estimation of FRIs over space and time is complex and, depending on the approach used, estimates of FRI may not always be directly comparable. Intervals prior to the first and following the last fire on record cannot be known, and censoring is required, but not always used, to account for this (Fernandes et al. 2012). Analyses also differ in whether or not they weight each observation point associated with an FRI by the size of the unit burnt. These differences can substantially affect estimates of median FRI and estimates of the degree to which fire is dependent on a build-up of fuel (Table 3.1, Fernandes et al. 2012; Kraaij, Baard, et al. 2013). Future studies should use both censoring and area weighting.

The question of whether fynbos requires relatively long periods between fires to build up sufficient fuel to support a subsequent fire is an important one with implications for fire risk management. Fuel-age dependency may be estimated by the Weibull shape parameter c (Table 3.1, Moritz, 2003). Fire hazard is age-independent when $c = 1$, increases linearly with fuel age when $c > 1$, and increases exponentially when $c > 2$ (Fernandes et al. 2012). Although fuel-age dependency may have been underestimated in fynbos where FRI analyses have not incorporated area weighting (e.g. van Wilgen et al. 2010), the reported values for c (typical values of 1.2–1.5) in fynbos imply that young vegetation is not fireproof (Table 3.1). Field observations confirm that fynbos may burn from three to five years of age under suitable conditions (van Wilgen, Everson, et al. 1990; Brown et al. 1991). Fuel-reduction burning of fynbos therefore would not necessarily provide reliable barriers to fire spread, although strategic placement of frequently burnt firebreaks may be useful for containing wildfires (Moritz 2003).

3.3.2 Fire season

Fynbos fires occur throughout the year, but are concentrated in the dry summer (November–March) in most of the CFR (Seydack et al. 2007; van Wilgen et al. 2010). In the far eastern coastal zone, the fire regime is largely aseasonal (Kraaij, Baard, et al. 2013), reflecting reduced seasonality in climate controls (Kraaij, Cowling, and van Wilgen 2013a). Weather conditions are comparatively mild (fire danger peaks in winter), and fires occur when aseasonal lightning is followed by elevated FDI (Kraaij, Cowling, and van Wilgen 2013a). In this zone, natural and human-ignited fires were equally aseasonal (Kraaij, Baard, et al. 2013), whereas in the eastern inland zone, fires of human origin were less strongly concentrated in summer than lightning-ignited fires (Seydack et al. 2007). Compared to wildfires, prescribed fires were historically overrepresented during the cooler months in the western and southwestern coastal, and western inland zones (Brown et al. 1991; van Wilgen et al. 2010); underrepresented during winter (due to restrictions on burning during the berg wind season) in the eastern coastal zone (Kraaij, Baard, et al. 2013); and shifted from late summer to autumn at the extensively burnt Bontebok National Park (Kraaij 2010). Overall, human interference is unlikely to have substantially altered fire seasonality, given the overriding importance of climatic controls and small contribution of prescribed burns to the total area burnt (van Wilgen et al. 2010).

3.3.3 The cause of fires

The cause of fires is often unknown, with up to 63% of the area burnt on record having burnt in fires of unknown origin (in the case of Table Mountain National Park; Forsyth and van Wilgen 2008). Natural causes of fynbos fires include ignitions by lightning strikes and less commonly, falling rocks during scarp retreat or earth tremors (Kruger and Bigalke 1984). Lightning fires dominate the fire regimes of remote mountain catchment areas such as the Swartberg, Kammanassie, and Tsitsikamma Mountains, while human ignitions are more important in more densely settled areas (Table 3.1). Fynbos fire regimes are dominated by unplanned wildfires (of natural or human origin), which account for almost 90% of the area burnt (van Wilgen et al. 2010).

3.3.4 Fire size

Fire size may be important for a number of reasons, such as the relative proportion of edge effects, or the distances that must be covered by species reliant on dispersal for recolonization of burnt sites. Fynbos fire regimes are typically dominated by few, very large fires. Large numbers of small fires do occur, but do not contribute much to the total area burnt (Table 3.1). This relationship between the number and size of fires is characteristic of many vegetation types globally (Keeley et al. 1999, 2012; Gill and Allan 2008; van Wilgen et al. 2010; Kraaij, Baard, et al. 2013). The largest fynbos fire on record burnt 58 528 ha in the Cederberg in 1988 (Keeley et al. 2012) and the next largest, 41 902 ha in the Tsitsikamma in 1998, with only 29 of 3 227 fires on record having exceeded 10 000 ha (van Wilgen et al. 2010; Kraaij, Baard, et al. 2013). Large fires appear to have become more common in recent times, with fire size distributions remaining stable over time in only two of eleven fynbos protected areas examined (Brown et al. 1991; Forsyth and van Wilgen 2007; Kraaij, Baard, et al. 2013).

Fire size is positively correlated with fire duration; in the eastern coastal zone, fires of <1000 ha burnt for an average of <2 days, fires of 1 000–10 000 ha burnt for 2–7 days, and fires of $\geq 10\ 000$ ha burnt for more than 7 days (maximum 40 days; T. Kraaij unpublished data). Fire size is, furthermore, positively correlated with elevated fire danger weather conditions (see 3.2.2). Fires were largest on average during summer and autumn in the eastern inland zone (Seydack et al. 2007), and in spring and autumn in the eastern coastal zone (Kraaij, Baard, et al. 2013) in line with fire climate seasonality.

3.3.5 Fire intensity and severity

Fire intensity refers to the rate at which energy is released during combustion, and different metrics of fire intensity (e.g. reaction intensity, fire-line intensity, heating duration, and radiant energy) are useful for different purposes (Keeley 2009). Fire intensity varies with the amount of fuel per unit area, and the time it takes to burn. Fire intensity is related both to the ease of controlling a fire, and to the effects on the biota. Shrublands do not have high fuel loads compared to forests, but the proportion of fuel consumed is relatively high due to their fine, readily combustible fuels. Differences in the vertical structure of plants and dead plant material, rather than fuel loads or fuel chemistry, often account for differences in fire behaviour between vegetation types (e.g. forest versus fynbos (van Wilgen, Higgins, et al. 1990) or chaparral versus fynbos (van Wilgen and van Hensbergen 1992)).

Attempts to quantify fire intensity in fynbos ecosystems have focussed on Byram's (1959) measure of fire-line intensity, estimated as the product of heat yield, the mass of fuel combusted and the rate of spread of the head fire front. Rates of fire spread in fynbos observed during small experimental plot burns (van Wilgen et al. 1985) or in prescribed fires (van Wilgen and McDonald 1992) ranged from 0.1–1.4 ms^{-1} , flame lengths ranged from 3–7 m, and fire-line intensity ranged from 500–20 000 kWm^{-1} . In 12-year-old fynbos, under the hot and dry conditions prevailing during large wildfires, Rothermel's model predicted a rate of spread of 3.6 ms^{-1} , fire-line intensity of 47 000 kWm^{-1} , and flame length of 5 m, decreasing to <1 m under conditions of low fire danger (van Wilgen et al. 1985; van Wilgen, Higgins, et al. 1990). Under comparable conditions of wind and fuel moisture, fires in fynbos will burn at a lower intensity than those in chaparral (van Wilgen 1984b), but at a higher intensity than those in grasslands, savanna or afrotemperate forest (van Wilgen, Higgins, et al. 1990; van Wilgen and van Hensbergen 1992).

Keeley (2009) differentiates between fire intensity, fire or burn severity (above-ground and below-ground organic matter consumption from fire), and ecosystem responses (functional processes that are altered by fire including regeneration, recolonization by plants and animals and hydrological processes altered by fire). In fynbos ecosystems, fire or burn severity has seldom been formally quantified. There is some work on soil water repellency (Scott and van Wyk 1992; van Wilgen and Scott 2001) showing that increases in biomass due to invasion by alien shrubs and trees increase fire

Table 3.1 Fire regime characteristics of fynbos protected areas (ordered from north to south for western zones and from west to east for others) in the Cape Floristic Region.

	Cedarberg ^{1,2,3}	Watrerval ^{2,3}	Nuweberg ^{2,3}	Limietberg ^{2,3}	Table Mountain ^{2,3,4}	Kogelberg ^{2,3}	Riversonderend ^{2,3}	Bontebok ⁵	Swartberg ^{2,3,6}	Kamansassie ^{2,3}	Western Outeniqua ^{2,3}	Eastern Outeniqua ⁷	Tsitsikamma ⁷
Fire climate zone	Western inland	Western inland	Western inland	Western inland	Western coastal	Western coastal	Southwest coastal	Southwest coastal	Eastern inland	Eastern inland	Southwest coastal	Southeast coastal	Far eastern coastal
Latitude (°S)	32.50	33.35	34.12	33.52	34.15	34.27	34.00	34.03	33.35	33.58	33.87	33.84	33.91
Longitude (°E)	19.00	19.08	19.13	19.15	18.38	18.93	19.57	20.42	22.32	22.85	22.60	22.80	23.85
Protected area size (ha)	122 735	108 055	59 936	90 044	26 554	37 851	69 046	3435	120 416 ² 170 856 ⁶	49 591	41 962	22 590	87 429
Mean annual rainfall (mm)	400–1700	420–440	700–780	530–570	400–1700	700–1100	590–600	528	100–1000	300–700	280–620	820	1078
Fire record	1956–1986 ¹ ; 1970–2006 ² ; 1945–2006 ³	1974– 2007	1972– 2007	1970– 2005 ² ; 1966– 2006 ³	1970–2007	1970– 2007 ² ; 1953– 2007 ³	1970– 2007	1972–2009	1970–2006 ² ; 1944–2006 ³ ; 1930–2002 ⁶	1970–2006	1970–2006	1900–2010	1900–2010
Number of fires	115 ²	182 ²	195 ²	220 ²	373 ²	98 ²	110 ²	43 ²	115 ² ; 506 ⁵	75 ²	162 ²	673 (cause); 107 (FRI)	766 (cause); 75 (FRI)
Mean or median fire return interval (yr):	11–15 a; ¹ 13 c ²	12 c ²	11 c ²	10 c ²	12 c ²	16 c ²	10 c ²	7.2 a	21 c ² ; Xeric: 30–55 a ⁶ Mesic: 15–30 a ⁶	11 c ²	18 c ²	11	7
c = censored												13 a	8 a
a = area-weighted												17 c	23 c
Weibull shape parameter c	1.4 c ²	1.5 c ²	1.2 c ²	1.4 c ²	1.2 c ²	1.5 c ²	1.5 c ²		1.3 c ²	1.5 c ²	1.5 c ²	1.9	2.0
c = censored												2.5 a	3.2 a
a = area-weighted												1.8 c	1.2 c
												2.5 ca	1.4 ca
% Area burnt per fire cause:													
Natural	43 ³	28 ³	15 ³	4 ³	1 ³	17 ³	3 ³	0 ³	54/40 ^{3,6}	50 ³	15 ³	26	83
Prescribed	10	5	7	11	15	20	32	70	0/0	4	19	13	<1
Other human	35	18	24	26	21	17	57	0	6/36	3	49	34	10
Unknown	12	49	54	59	63	46	8	30	40/24	43	17	27	7
% Area burnt per size class:													
Small (0–100 ha)	1 ³	2 ³	2 ³	2 ³	13 ³	1 ³	1 ³	3 ³	2 ³	1 ³	3 ³	3	2
Medium (>100–2000 ha)	23	29	46	22	67	38	36	97	26	27	46	27	13
Large (>2000–5000 ha)	15	21	12	37	20	41	26	n/a	39	25	31	28	11
Very large (>5000 ha)	61	48	40	39	0	20	37	n/a	33	47	20	42	74

Sources: ¹Brown et al. 1991; ²van Wilgen et al. 2010; ³Forsyth and van Wilgen 2007; ⁴Forsyth and van Wilgen 2008; ⁵Kraaij 2010; ⁶Seydack et al. 2007; ⁷Kraaij, Baard, et al. 2013.

severity and soil damage. Breytenbach (1989) quantified fire severity in various ways (loss of water from open beer cans, the depth at which plant tissue was killed, impacts on termite mounds, post-fire soil loss, and the degree of post-fire recovery) to demonstrate the negative effects of felling and burning dense stands of invasive alien *Hakea* shrubs.

3.3.6 Fire regimes in vegetation types other than fynbos

Apart from rainfall and soils, fire is a key determinant of the distribution of fynbos and other vegetation types within the CFR (Chapter 1, Keeley et al. 2012). Fynbos and renosterveld are fire-prone shrublands, fynbos occurring over a wide rainfall gradient on sandy, nutrient-poor soils and renosterveld on finer, richer soils in more arid sites. Forest and thicket (or 'strandveld' along the coast) occur on similar soils and under similar rainfall as fynbos or renosterveld, but in fire refugia, these being fire-resistant systems that seldom burn. Succulent karoo occurs on clay-rich soils with low rainfall and is fire free on account of its succulent nature and low biomass.

Renosterveld

Compared to fynbos, pre-colonial renosterveld would have been more heavily grazed by native mammals owing to its grassy nature and richer substrates. Fuel dynamics would have been affected by both fire and grazing, which would have impacted on fire regimes. The modern, drastically altered situation arose before any studies of this ecosystem could be undertaken. Less than 4% of coastal renosterveld remains and it is severely fragmented (Rebelo et al. 2006). The once extensive populations of large mammals have been exterminated. Consequently, little is known about the fuel dynamics and fire ecology of renosterveld. Renosterveld is thought to burn more frequently (FRI = 3–5 yr) than fynbos (FRI = 10–25 yr) due to faster plant growth and finer fuels (Boucher 1995; Rebelo et al. 2006). Amongst the few recent accounts of fire in renosterveld is that of the intensively managed Bontebok National Park (the vegetation being a mixture of lowland fynbos and south coast renosterveld) where the mean FRI was *c.* 6–7 yr during a time (1972–2009) when prescribed burning and intensive grazing were applied (Kraaij 2010). Under such a regime of intensive grazing, 6 yr appeared to be the minimum post-fire age required to allow sufficient fuel accumulation to support a fire under weather conditions suitable for prescribed burning. In the Table Mountain National Park

(where prescribed burning and intensive grazing were not practiced), mean FRIs in renosterveld were 37 yr in the 1970s and 18 yr during the period 2000–2007, the increase in fire frequency attributed to increased human ignition sources over time (Forsyth and van Wilgen 2008). There is also concern that a lack of fire and herbivory in isolated renosterveld fragments leads to woody shrub encroachment (Krug et al. 2004; van Wilgen et al. 2012). From a plant conservation perspective, FRIs of 10–15 yr are deemed appropriate in wetter renosterveld types, and longer cycles in drier, inland types and quartz–silcrete outcrops (Helme and Rebelo 2005; Curtis et al. 2013).

Strandveld and dune fynbos-thicket

Fire is not a principal ecological driver in strandveld as it is in fynbos and renosterveld. The frequency of fires in strandveld in the past remains largely unknown (Liengme 1987) but fires are rare (return intervals of 50–200 yr; Rebelo et al. 2006); the succulent nature of strandveld impeding the spread of fires and reducing their intensity (Rebelo et al. 2006). Little deliberate burning occurs in strandveld today (Liengme 1987). The season of fire could be critical, given the highly seasonal phenology of strandveld, as well as the intensity of fire, since excessively intense fires (often fuelled by alien plant material) may kill resprouters (Helme 2005). Strandveld will in all likelihood only burn in summer when the vegetation is dry, but our understanding of the 'natural' fire regime is minimal. In the more arid west coast regions, fire is hardly ever a factor since fuel loads are too low. Strandveld on dunes (coastal dune thicket) also does not burn readily, and coastal dune systems are particularly vulnerable to wind erosion when the vegetation cover is disturbed (Helme 2005).

The dynamic relationship between strandveld and adjacent fire-prone vegetation types is poorly understood. The responses of plants to herbivore pressure and recurrent fire influence the nature of the vegetation and may cause boundary shifts between these vegetation types (Liengme 1987). Fire is thus deemed important for the maintenance of the ecotone between strandveld and sandplain fynbos, as is the case with the more mesic dune fynbos–thicket mosaics found along the Cape south and south east coasts (Helme 2005). In these dune fynbos–thicket mosaics, dune thicket typically occupies fire-protected sites (e.g. calcrete outcrops or the northern base of dunes) while more combustible dune fynbos is associated with fire-prone, usually wetter, locations (Helme 2005). Dune fynbos–thicket is subject, on average, to summer–autumn fires

at intervals of 10–20 yr, although extremes of 4–40 yr are also possible (Pierce and Cowling 1991), with all components of the fire regime showing considerable variation (Cowling 1984). Persistence of dune fynbos–thicket requires fire at appropriate intervals (15–25 yr) since it is replaced by subtropical thicket in the prolonged absence of fire (Cowling 1984).

Forest

Afrotropical forest in the CFR typically occurs within a matrix of fire-prone fynbos, and is fire free or, at least, fire resistant (Geldenhuys 1994). Although forests are unlikely to burn under conditions favouring fire in the surrounding fynbos (van Wilgen, Higgins, et al. 1990), soils in southern Cape forests contain charcoal indicating that they have occasionally burnt (Watson and Cameron 2002). Fire plays an important role in the dynamics of forest–fynbos margins and smaller mountain forest patches, as evidenced by differences in the composition, diversity, and mortality of tree species in the margin and core of forests (Watson and Cameron 2001). Unlike the larger plateau forests, mountain forests are thought to burn more frequently (every 100–200 years; Watson and Cameron 2002). Fire-resistant, resprouting species, such as *Ocotea bullata* and *Cunonia capensis*, dominate forest margins, largely surviving the relatively intense fires penetrating forest margins. In contrast, the lower intensity of fires in forest cores allows a more diverse community of fire-sensitive species to survive (Watson and Cameron 2001).

3.4 Responses to fire

3.4.1 Soils, water, and nutrients

Plants affect the hydrological cycle through rainfall interception and transpiration. Fires alter this by periodically removing plant cover and triggering plant regrowth. Post-fire increases in streamflow attributed to reductions in rainfall interception and transpiration range from 70–120 mm yr⁻¹ or 12–133% during the first two years post fire (Scott and van Wyk 1992) but decline with increasing vegetation age and vary according to vegetation structure and composition (Bosch et al. 1986).

Overland stormflows, sediment yields, and soil water repellency may be unaffected or increase after fire (62% increase in stormflow, <0.001–0.416 tonnes ha⁻¹ yr⁻¹ increase in sediment yields), but rapid recovery to pre-fire levels is typical (Scott and van Wyk 1992). Soil water repellency results when plant

materials form hydrophobic, organic substances which coat soil particles by leaching downwards or by vapour transport during burning. Water repellency decreases as organic matter content of soils decreases and with increased soil heating. High fire intensities associated with excessive fuel loads caused by afforestation or invasion by alien trees may have more severe hydrological effects (e.g. 290–1110% increases in peak discharge; 6 tonnes ha⁻¹yr⁻¹ sediment yield; Scott et al. 1998).

Nutrients released by fire will vary with site, fire intensity, and the extent of fuel consumption (van Wilgen and Le Maitre 1981; De Ronde 1990). Fire in coastal fynbos increased total N and NH₄⁺-N concentrations at the soil surface immediately post fire, the latter being short lived (Stock and Lewis 1986). Indirect effects of fire were apparent on soil N form, distribution, and concentration in the year following fire. Total N concentrations increased at greater depths and decreased at the surface, due to leaching (Stock and Lewis 1986). Available soil NO₃-N showed a nine-month post-fire flush due to increases in bacterial nitrifiers or decreased NO₃-N demand resulting from removal of the mature vegetation (Stock and Lewis 1986). High-intensity fires may furthermore significantly reduce above-ground P, K, and Ca through volatilization, increasing their levels in topsoil (De Ronde 1990). Nutrient losses through streamflow following fire are small (van Wilgen and Le Maitre 1981) and do not persist beyond the first winter post fire, largely occurring during the first two post-fire floods (van Wyk et al. 1992). Volatilization losses of mineral nutrients in fires may significantly affect the nutrient status of catchments, although continual outputs through stream water far exceed losses through irregular fire events. The effects of alien *Acacia* infestations on soil chemistry also considerably exceed that of fire (Musil and Midgley 1990). Post-fire regrowth and unburnt riparian zones play an important role in acting as nutrient filters or traps (van Wyk et al. 1992). Overall, fire is an effective mineralizing agent in fynbos (Stock and Lewis 1986) but has little long-term effect on the nutrient budgets of fynbos catchments under normal fire frequencies (van Wyk et al. 1992).

3.4.2 Fynbos reproduction and persistence

Fire survival and persistence

Fynbos plants exhibit a wide range of regeneration strategies and fire-survival mechanisms (Table 3.2). Whether or not particular plant life history traits are evolutionary adaptations (or 'exaptations') to fire, has been extensively debated in recent literature

(Bradshaw et al. 2011a, b; Keeley et al. 2011; Midgley and Bond 2011; Keeley et al. 2012; Pausas and Schwillk 2012). Most fynbos species can resprout after fire, although few are obligate resprouters (Musil and De Witt 1990; van Wilgen and Forsyth 1992). Only a small proportion of species are obligate reseederers, many of which are in the Proteaceae, Ericaceae, and Fabaceae (Le Maitre and Midgley 1992; Schutte et al. 1995). Most (86%) fynbos plants flower within one year of a fire and only 2% of species (largely the Proteaceae) take longer than 3 yr to flower (van Wilgen and Forsyth 1992). Most species (>80%) are long-lived (>45 yr), 10% are short lived (<10 yr), and there are few annuals. Short-lived species, such as some *Aspalathus*, *Roella*, and *Thesium* species, are assumed to have fairly long-lived seedbanks, enabling them to reappear after fire (van Wilgen and Forsyth 1992). The relative species richness and cover abundance of life history types varies widely within and amongst mediterranean-climate ecosystems (Le Maitre and Midgley 1992; Keeley et al. 2012). The prevalence of reseederers versus resprouters furthermore varies in response to fire interval and intensity (Bosch et al. 1986; Musil and De Witt 1990; van Wilgen and Forsyth 1992; Vlok and Yeaton 2000a). Post-fire recruitment in both is more important than inter-fire recruitment, the latter being possible but uncommon (Bond 1980; Le Maitre 1992). Reseederers often accumulate larger seedbanks than ecologically similar resprouters (Le Maitre 1992), but in west coast sandplain fynbos, post-fire seedling production in reseederers and resprouters was similar (Musil and De Witt 1990). Large-seeded myrmecochorous reseeding perennials emerged sooner post fire and from greater depths, producing more seedlings than small-seeded reseederers (Musil and De Witt 1990).

In mountain fynbos, woody serotinous obligate reseederers (almost exclusively members of Proteaceae; hence hereafter 'proteoids') often comprise only a few species but can make up 90% of the mature biomass (van Wilgen and Forsyth 1992). Post-fire regeneration from seed in these proteoids is highly variable in space and time (Figs 3.4 and 3.5, van Wilgen and Viviers 1985; Le Maitre 1988a; Cowling and Gxaba 1990). Populations are decimated or locally extirpated if fire cycles are shorter than the juvenile period or if post-fire conditions are unfavourable for seedling recruitment, because the entire short-lived seedbank is released post-fire (Lamont et al. 1991) and germinates quickly (Midgley et al. 1989; Kraaij 2012). Variation in recruitment success has been explained on the basis of: seed age (Le Maitre 1990), the size of pre-fire seedbanks which vary with plant age (Fig 3.6, Bond 1980;

Le Maitre 1990; Lamont et al. 1991; Kraaij, Cowling, et al. 2013), plant phenology, pre-fire density (Bond et al. 1984; Le Maitre 1988a; Cowling and Gxaba 1990; Esler and Cowling, 1990), pre-dispersal seed predation (Esler and Cowling 1990), post-dispersal seed predation and decay (and thus duration of seed exposure between release and germination; Bond 1984; Bond and Breytenbach 1985; Botha and Le Maitre 1992; Le Maitre 1994), climatic conditions favourable to germination (requirement for cold, moist conditions; Le Maitre 1990), soil type (Mustart and Cowling 1993), presence of alien plant infestations (Musil 1993), and the extent of post-germination mortality due to fungal pathogens, vertebrate and invertebrate herbivory, highly individualistic levels of desiccation tolerance, and density-dependent thinning (Midgley 1989; Midgley et al. 1989; Botha and Le Maitre 1992; Bond et al. 1995; Mustart et al. 2012).

Fire season effects

Fire season can have marked effects on post-fire seedling recruitment in those zones where fires are concentrated in summer (Fig 3.1). Flowering in the western regions of the CFR peaks in spring and reaches a trough in autumn and early winter, which implies that maximum seed loads will be available in late summer or early autumn (Johnson 1992). In the mediterranean-climate shrublands of the CFR and Australia, recruitment of serotinous proteoids is accordingly higher after fires in summer–autumn than those in winter–spring (van Wilgen and Viviers 1985; Le Maitre 1988a; Midgley 1989). Proteaceae with soil-stored seed also may not recruit after fires in spring, which is prior to maturation and shedding of the current year's seed crop (Le Maitre 1988b). Flower and seed production in geophytes exhibiting fire-stimulated flowering (e.g. *Watsonia borbonica*) may also be reduced by foliage removal (fire) during winter or early spring as opposed to summer or early autumn (Le Maitre and Brown 1992). Recruitment of pioneer shrubs like *Anthospermum aethiopicum*, *Stoebe plumosa*, and *Metalasia muricata* seems less affected by fire season (Jordaan 1982). In the eastern coastal CFR, flowering peaks in early summer (Johnson 1992), fire seasonality and intensity are variable (van Wilgen 2009; Kraaij, Cowling, and van Wilgen 2013a) and the effects of fire season on proteoid recruitment are less pronounced (Fig 3.4, Heelemann et al. 2008; Kraaij 2012). Neither plant phenology (Johnson 1992; Heelemann et al. 2008) nor rainfall imposes seasonal constraints on proteoid recruitment in this area (Kraaij 2012), implying that season of burn is essentially unimportant. However,

Table 3.2 Examples of the major fire survival and persistence strategies in fynbos vegetation.

Fire survival or persistence strategy	Mechanism	Examples
Species that rely solely on reproduction by seed to survive fires (obligate reseeders)—serotiny	Retention and protection of seeds in persistent fruits for overlapping seasons. Entire seedbanks released post fire following the death of parent plants (lacking resprouting capacity). Seeds short lived after release, mostly germinating during first favourable period. Seeds do not require fires or post-fire environmental conditions for germination and germinate best under cold conditions (<15°C).	<i>Protea</i> , <i>Leucadendron</i> , <i>Erica sessiliflora</i> , <i>Widdringtonia cedarbergensis</i> , <i>W. schwarzii</i>
Obligate reseeders—ant-(myrmecochory) or rodent-dispersed	Burial of large, protein-rich seeds (often with eliasome) by ants or rodents. Fire stimulates germination of hard-coated soil-stored seeds through scarification, charate, or by providing appropriate soil temperature regimes for germination.	c.1300 spp. (20% of strictly fynbos spp.) in 29 families: 168 Proteaceae spp. (<i>Mimetes</i> , <i>Leucospermum</i> , <i>Serruria</i> , <i>Orothamnus zeyheri</i>), some Restionaceae (e.g. <i>Cannomois virgata</i>), Rutaceae, many species of Fabaceae and Ericaceae (e.g. <i>Erica hispidula</i> , <i>E. longifolia</i> , <i>E. sphaeroidea</i>), <i>Chrysanthemoides monilifera</i> , <i>Aristea africana</i> , <i>Sebaea exacoides</i> , <i>Chironia baccifera</i> , <i>Phylca pubescens</i> , <i>Cliffortia cuneata</i> , <i>Cliffortia polygonifolia</i> , <i>Anthospermum galioides</i> , <i>Solanum tomentosum</i> , <i>Struthiola ciliata</i> , <i>Helichrysum cymosum</i> , <i>Metalasia muricata</i>
Facultative resprouter—reseeders	Species able to reproduce by vegetative means (initiation of new shoots from existing meristems) and sexually (from seed).	54 Proteaceae spp. incl. <i>Protea nitida</i> , <i>Leucadendron salignum</i> , <i>Diospyros</i> , <i>Heeria</i> , <i>Seersia</i> , <i>Maytenus</i> , <i>Myrsine</i> ; Geophytes (Iridaceae, Orchidaceae, Hyacinthaceae, Amaryllidaceae, Hypoxidaceae, Oxalidaceae); most Cyperaceae, Poaceae, Apiaceae, Bruniaceae, Campanulaceae, <i>Corymbium</i> spp., <i>Helichrysum nudifolium</i> , <i>H. teretifolium</i> , <i>H. zeyheri</i> , <i>Phylca imberbis</i> , <i>Gnidia oppositifolia</i> , <i>Widdringtonia nodiflora</i>
Obligate vegetative resprouters	Species present after fire solely as vegetative resprouts with no seedling recruitment at the time.	<i>Acrolophia capensis</i> , <i>Athrixia heterophylla</i> , <i>Disparago ericoides</i> , <i>Clusia rubricaulis</i> , Restionaceae, e.g. <i>Calopsis membranaceus</i> , <i>Cannomois virgata</i> , <i>Elegia juncea</i> , <i>Hyposidscus aristatus</i> , <i>Ischyrolepis capensis</i> , <i>Restio triticeus</i> , <i>Staberoha cernua</i> , <i>Thamnochortus fruticosus</i> , <i>Willdenowia sulcata</i>
Heat-stimulated germination from soil-stored seed	Fire intensity needs to be sufficiently high to stimulate germination in hard-coated, soil-stored seeds.	<i>Podalyria calytrata</i> , <i>Erica hebecalyx</i>
Smoke-stimulated germination	Species in which exposure to a butenolide (van Staden et al. 2004) present in plant-derived smoke (charate) provides germination cues.	Large proportion of the fynbos flora, e.g. <i>Cannomois virgata</i>
Fire-stimulated flowering	Profuse flowering for one or more years after fire, largely due to indirect effects of fire (changes in the environment e.g. altered soil temperatures, nutrient availability) rather than direct effects (i.e. heat-induced damage to leaves or apical buds, or smoke-derived chemicals).	Most geophytes; fire lilies (<i>Cyrtanthus ventricosus</i> , <i>C. guthrieae</i> , <i>Haemanthus rotundifolius</i> , <i>Nerine sarniensis</i>); Iridaceae (<i>Watsonia borbonica</i>); Orchidaceae (<i>Disa bivalvata</i> , <i>D. fasciata</i> , <i>D. racemosa</i> , <i>D. tenuifolia</i>); <i>Xiphotheca rosmarinifolia</i> ; many herbaceous species, e.g. <i>Ehrharta capensis</i>
Fire ephemerals	Ephemerals, annuals, or short-lived perennials which exhibit a degree of fire-stimulated germination.	<i>Senecio pubigerus</i> , <i>Ursinia paleacea</i> , <i>U. pinnata</i> , <i>Roella ciliata</i> , <i>Aspalathus ciliaris</i> , <i>Polygala bracteolata</i> , <i>Thesium capitatus</i> , <i>T. strictum</i> , <i>T. virgatum</i>
Fire avoiders	Species that grow in fire refugia such as rocky scarps, where fires seldom penetrate. Seedling reproduction is not fire dependent.	Many trees (e.g. <i>Heeria argentea</i> , <i>Maytenus oleoides</i> , <i>Podocarpus elongatus</i>); <i>Aloe mitrifolius</i>

Sources: van Wilgen 1980; Brits 1986; Jeffery et al. 1988; Midgley and Viviers 1990; Musil and De Witt 1990; Le Maitre and Brown 1992; Le Maitre and Midgley 1992; Le Maitre et al. 1992; van Wilgen and Forsyth 1992; Schutte et al. 1995; Johnson and Bond 1997; Watson and Cameron 2001; Verboom et al. 2002; Brown and Botha 2004; Newton et al. 2006; Schutte-Vlok 2011; Keeley et al. 2012; Rusch et al. 2013.

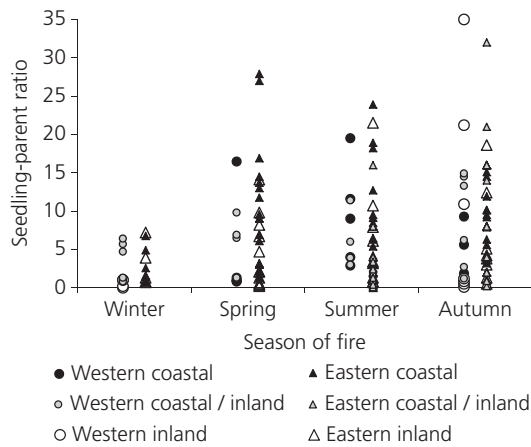


Figure 3.4 Recruitment success of reseeded serotinous Proteaceae after fires in different seasons and in different climatic zones (see Fig 3.1 for delineation). Sources: Western coastal, Kruger and Lamb 1979; Le Maitre 1988a; Cowling and Gxaba 1990; Western inland, Jordaan 1965, 1982; Le Maitre 1992; Western coastal/inland, van Wilgen and Viviers 1985; Eastern coastal, Midgley 1989; Kraaij 2012; Eastern inland, Bond 1980; Heelemann et al. 2008; Eastern coastal/inland, Bond 1984; Bond et al. 1984.

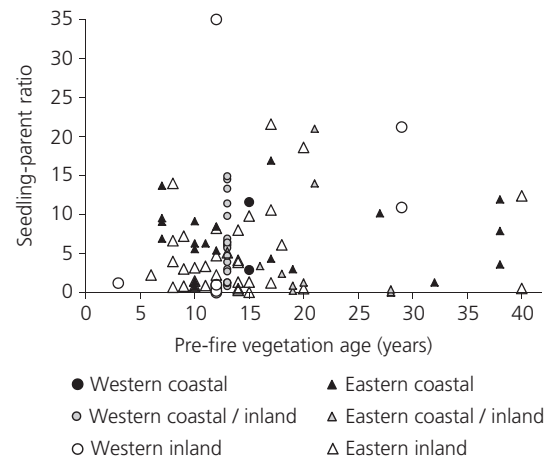


Figure 3.5 Post-fire recruitment success of reseeded serotinous Proteaceae in different climatic zones (see Fig 3.1 for delineation) in relation to the age at which the vegetation burnt. Sources: Western coastal, Cowling and Gxaba 1990; Western inland, Jordaan 1965, 1982; Le Maitre 1992; Western coastal/inland, van Wilgen and Viviers 1985; Eastern coastal, Kraaij, Cowling, et al. 2013; Eastern inland, Bond 1980; Heelemann et al. 2008; Eastern coastal/inland, Bond et al. 1984.

throughout the CFR, autumn fires mostly result in good recruitment of fynbos (Midgley 1989).

Fire frequency effects

The life histories of obligate reseeded shrubs are strongly influenced by fire frequency, as they need sufficient time to mature and set seed between fires. Estimates of ecologically optimal fire frequencies are consequently based, at the lower end, on juvenile periods of the slowest-maturing obligatory reseeded species, and at the higher end, on the longevity of short-lived reseeded species that do not maintain long-lived seedbanks. In fynbos, the proteoids are indicators at both ends of the spectrum. The effects of fire on this group have accordingly been well studied (Figs 3.4 and 3.5) while little or nothing is known for many other groups of plants (e.g. the genus *Erica*, in which almost all of the 658 species are thought to rely on regeneration from seed (van Wilgen and Forsyth 1992)). The rule of thumb generally used in fynbos to establish minimum FRI posits that half of the individuals in a population of the slowest-maturing proteoid species should have flowered over at least three successive seasons before the area may be burnt (Kruger and Lamb 1979). Proteoid juvenile periods range from 4–9 yr (Fig 3.6) and are better correlated with plant size (e.g. height) than with plant age (Kruger and Bigalke 1984; Le Maitre 1992; Kraaij, Cowling, et al. 2013). Juvenile periods are

generally comparable across regions of the CFR, but perhaps somewhat shorter in coastal than in inland areas. Variation in juvenile periods appears related to local moisture regimes (i.e. shorter in moister habitats where plant growth rates are faster (Le Maitre and Midgley 1992; Seydack et al. 2007; Kraaij, Cowling, et al. 2013)).

Short FRIs (<7yr; Fig 3.5, Kraaij, Cowling, et al. 2013) thus reduce or eliminate obligate reseeded proteoids from the vegetation, while resprouting shrubs are less affected (although frequent fires also cause a slow attrition of resprouter populations; Le Maitre 1992), while resprouting graminoids (Restionaceae and Cyperaceae) proliferate (Vlok and Yeaton 2000a). The presumption that prescribed burning is required to prevent vegetation senescence and loss of short-lived proteoids lacks evidential support, as wildfires provide ample opportunity for rejuvenation (van Wilgen 2009). Apart from the proteoids, most plant species are resilient to a wide range of FRIs (van Wilgen and Forsyth 1992). The extent of post-fire survival of woody resprouting species partially depends on pre-fire plant size, and the abilities of plants to tolerate drought (Le Maitre et al. 1992). Resprouting species with high fire-induced adult mortality produced more seedlings than those with low adult mortality (Le Maitre and Midgley 1992; Le Maitre et al. 1992). Furthermore, very short FRIs (<4 yr) may adversely affect geophytes which need to attain a 'critical mass' of reserves before they will

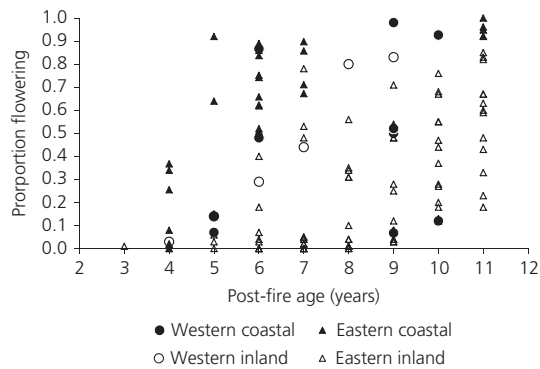


Figure 3.6 Proportions of populations of reseeding serotinous Proteaceae that have flowered at least once at different post-fire vegetation ages. Sources: Western coastal, Kruger and Lamb 1979; Western inland, Le Maitre 1992; Eastern coastal, Kraaij, Cowling, et al. 2013; Eastern inland, J.H.J. Vlok (unpubl. data).

flower (Le Maitre and Brown 1992). Van Wilgen and Forsyth (1992) suggested that fynbos communities should mostly be maintained by FRIs of 10–20 yr, although Schwilk et al. (1997) found that species diversity peaked at sites where FRIs were 40 yr, and were lowest at sites of moderate (15–26 yr FRIs) and high fire frequency (4–6 yr FRIs). Dry fynbos types, such as west coast sandplain fynbos, may also require relatively long FRIs (Hoffman et al. 1987).

Fire intensity effects

High-intensity fires facilitate (a) seed release and germination in serotinous species (Midgley and Viviers 1990); (b) scarification (and thus imbibition) of hard-coated soil-stored seeds (Jeffery et al. 1988; Bond et al. 1990); and (c) complete consumption of above-ground plant biomass causing changes in light and soil temperature regimes which stimulate germination of seeds of myrmecochorous and other species (e.g. *Leucospermum*, *Serruria*; Brits 1986). Conversely, high-intensity fires may kill small soil-stored seeds particularly when shallowly buried (Bond et al. 1999). Generally, short-rotation and low-intensity fires favour fast-maturing and resprouting species while long fire-free intervals (>30 yr) result in high-intensity fires favouring myrmecochorous Proteaceae and negatively impact serotinous Proteaceae (Vlok and Yeaton 1999, 2000a, b). Survival of adult Clanwilliam cedars (*Widdringtonia cedarbergensis*), a slow-growing reseed, may be compromised by intense wildfires (van Wilgen 1980). A remarkable example that illustrates the importance of fire intensity for shrub survival is provided by

Mimetes stokoei, which was thought to be extinct for several decades. Attempts to stimulate germination of soil-stored seeds by applying low-intensity prescribed burns failed to produce any germination. Seedlings did eventually emerge, however, following a high-intensity wildfire (Slingsby and Johns 2009).

The importance of variability

The composition of fynbos vegetation, while influenced by single fires, is really the product of multiple fires over time. Fynbos fire regimes are highly variable over space and time, and successive fires in the same area differ substantially in return intervals, season, intensity, and size. This variation can be important for long-term species coexistence, as particular species or guilds are not consistently favoured over others (Cowling and Gxaba 1990; Esler and Cowling 1990; Thuiller et al. 2007). Overstorey proteoids restrain the competitive ability of understorey resprouters post fire, and amplify within-community patchiness of understorey species, which ultimately enhances the species richness of fynbos communities (Vlok and Yeaton, 1999, 2000a, b). However, consistent burning under conditions optimal for post-fire recruitment of serotinous Proteaceae may, for instance, select for uniformly dense populations of these species, which may also suppress understorey diversity (Cowling and Gxaba 1990; Esler and Cowling 1990) or result in endogenous population crashes of proteoids subsequent to the next fire (Bond et al. 1995). Stochastic environmental fluctuations associated with recurrent fire buffer populations from extinction, thereby ensuring stable coexistence at the meta-community scale by maintaining patchiness through niche differentiation (Cowling 1987; Cowling and Gxaba 1990; Thuiller et al. 2007).

3.4.3 Renosterveld

Fire survival strategies and post-fire succession

Serotiny, which is a distinguishing feature of fynbos, is noticeably lacking in renosterveld. Higher proportions of resprouters and faster growth and maturation rates of reseeders suggest that renosterveld has greater resilience against short cycle fires than proteoid fynbos (Boucher 1983; Curtis 2013). Annuals, reseeding forbs, and shrubs depend on soil-stored seedbanks for post-fire reproduction, while geophytes, graminoids, and some shrubs survive fires by resprouting from below-ground perenniating organs. Almost 60% of species resprouted after fire, while 30% reseeded and 10% used both strategies (Curtis 2013). Seedling recruitment on

burnt sites was two to three times that on unburnt sites and most of the dominant shrubs (Asteraceae) only produced seedlings after fire (Curtis 2013). The role that obligate reseeders play in renosterveld ecosystems remains poorly understood, as well as the effects of high burning frequencies on slow-maturing, rare species (Curtis 2009; e.g. *Relhania garnotii*, which presumably takes 20 yr to mature; Curtis et al. 2013).

A study of post-fire succession of renosterveld in the Roggeveld Mountains showed rapid (within two years) re-establishment of the pre-fire community, with species diversity peaking within the first three years post fire and being at its lowest in old (9–10 yr) vegetation (van der Merwe and van Rooyen 2011). Seedling recruitment may occur in pulses, coinciding with above-average or consistent winter rainfall (Walton 2006; van der Merwe and van Rooyen 2011). Post-fire renosterveld vegetation is initially dominated by a ground stratum of herbaceous plants (annuals, geophytes, and hemicryptophytes) which is progressively replaced by dwarf shrubs (e.g. *Aspalathus*, *Hermannia*) and shrubs (*Elytropappus rhinocerotis*, *Athanasia trifurcata*, *Stoebe plumosa*), often forming two strata (Walton 2006; van der Merwe and van Rooyen 2011). Geophyte richness generally does not diminish with post-fire succession, similar to lowland fynbos, but unlike mountain fynbos (Walton 2006). *Elytropappus rhinocerotis* appears to have a facilitative role as a nurse-plant or refuge site protecting geophytes, forbs, hemicryptophytes, and dwarf shrubs from exposure or grazing and trampling (Walton 2006). Brush-cutting does not elicit the same response as fire in stimulating *E. rhinocerotis* germination; instead it increases the hemicryptophyte component, eventually selecting for those growth forms and resulting in structural degradation (Boucher 1995; Walton 2006). Post-fire successional patterns may change in the presence of exotic annual grasses which outcompete shrub seedlings and increase fire frequencies (Walton 2006). In disturbed sites where the alien annual forb *Echium plantagineum* is present, fire is likely to stimulate large-scale germination of this aggressive invader and compromise restoration attempts (Heelemann 2010).

Interactions between fire and herbivory

Fire and herbivory are the main ecological processes governing vegetation structure and diversity in renosterveld, but their relative importance is related to, and as contentious (Krug et al. 2004; Radloff 2008), as the debate about what renosterveld actually is, a grassy shrubland or shrubby grassland (Rebelo et al. 2006).

Renosterveld is thought to be less fire dependent than fynbos given that seeds of renosterveld species generally do not show adaptations to post-fire dispersal or exhibit smoke- or heat-induced germination (Krug et al. 2004; Midoko-Iponga 2004; Heelemann 2010). Yet, Curtis (2013) interpreted the high proportion of non-sprouters in renosterveld flora, and an increase in diversity, abundance, and flowering of plants after fire as evidence of a fire-driven system. Krug et al. (2004) argued that renosterveld is principally driven by herbivory, suggesting that the postcolonial decimation of indigenous herbivores would have resulted in increased fuel loads (in the absence of herbivory), causing hotter and larger-scale fires which would have homogenized the landscape towards shrublands (in particular *Elytropappus rhinocerotis*). On the other hand, severe and continuous overgrazing by domestic stock and the absence of fire both promote the dominance of asteraceous shrubs and C₃ bunch grasses at the expense of C₄ grasses, and are often held responsible for the presumed switch from (tussock) grassland to shrubland and impoverishment of diversity (Radloff 2008; Curtis 2013).

The interaction between fire and grazing and how it affects the relative abundance of grasses, shrubs, geophytes, and forbs in renosterveld has been studied extensively (Midoko-Iponga 2004; Luyt 2005; Radloff 2008; Curtis 2009; Kraaij and Novellie 2010; Novellie and Kraaij 2010; Watson et al. 2011). Herbivory by native grazers and browsers, or the release from it, cannot by itself bring about vegetation state changes in renosterveld, although a combination of herbivory and fire can have major effects on vegetation state dynamics through direct and indirect effects of herbivory on seedling survival after fire (Radloff 2008). At Bontebok National Park, all large herbivores (bulk grazers, concentrate grazers, browsers, and mixed feeders) preferred recently burnt vegetation, mostly avoiding vegetation >5 yr old (Kraaij and Novellie 2010); and despite a regime (1974–2009) of frequent burning (aimed at promoting grassiness) and intensive grazing, cover of *Themeda triandra*, a palatable tussock grass species, has not changed (Novellie and Kraaij 2010). However, the combination of fire and high grazing pressure may result in the initiation or spread of *Cynodon dactylon* grazing lawns (Luyt 2005; Radloff 2008) at the expense of floral diversity. Many of the non-toxic geophytes and annuals in renosterveld are furthermore vulnerable to grazing pressure by domestic stock in the first two years after fire (Helme and Rebelo 2005), while quartz-silcrete outcrops are sensitive to heavy trampling (Curtis et al. 2013).

3.4.4 Forest–fynbos boundaries

Fynbos is fire prone but contains fire-resistant forest patches which seldom burn (van Wilgen, Higgins, et al. 1990). Afrotemperate forest within the CFR is most extensive in the eastern coastal zone, occurring largely on the coastal plateau along the foothills of the mountains, and elsewhere in fire refugia such as sheltered ravines, stream banks, scree slopes, and berg wind shadows in the mountains (Geldenhuys 1994). Although fuel loads in mature (12 yr) fynbos (15 tonnes ha⁻¹) may be only half those in adjacent forest (35 tonnes ha⁻¹), the physical and chemical make-up of fuels favours fire in fynbos but not in forest (van Wilgen, Higgins, et al. 1990). Differences in the spatial distribution of fuels seem to be of overriding importance in determining the capacity of these vegetation types to support fire. Forests have concentrated litter layers with high bulk densities and sparse crowns and thus lack bridging fuels, while fynbos has vertically continuous fuel beds about 1–2 m deep (van Wilgen, Higgins, et al. 1990). The moisture contents of live foliage in fynbos (98–126%) are also lower than those in forest (167–216%), while the crude fat (0.4–10.3%) and energy (21 860 Jg⁻¹) contents of fynbos fuels are higher than those of forest fuels (1.0–5.7% and 20 703 Jg⁻¹), resulting in higher flammability of fynbos (van Wilgen, Higgins, et al. 1990).

During severe droughts, surface fires may occur in forests, where they kill trees by burning their root systems (van Wilgen, Higgins, et al. 1990). Hot, dry, katabatic berg winds can periodically desiccate most fuels to the extent that they are flammable. Forest distribution within the fynbos matrix has accordingly been explained in relation to topography and associated berg wind corridors, forest occurring in berg wind shadows and thus fire refugia (Geldenhuys 1994). Forest patches within fynbos are characterized by abrupt margins which are maintained by fire (Geldenhuys 1994). These margins may be scorched by fires, the degree of penetration into the forest depending on the intensity of the fire in the adjacent vegetation. Small forest patches or narrow riparian forests may be scorched entirely by intense fires in the adjacent fynbos, even though the patches themselves do not support fires (van Wilgen, Higgins, et al. 1990).

Most forest species can resprout after fire, but may be killed by high-intensity fires (Manders et al. 1992). Seed availability does not appear to limit post-fire colonization of fynbos by forest species, which disperse readily (by wind and birds) into fynbos after fire (Manders et al. 1992). Forest species may germinate under post-fire conditions in fynbos environments, but

survival of seedlings is restricted by high light intensities, low soil moisture, and low nutrients compared to forest environments (Manders et al. 1992). Between-fire development of forest in fynbos was unrelated to distance from the forest edge, but rather associated with tall vegetation (presence of bird perches), plant cover of >50% and a well-developed litter layer. While frequent fires in adjacent fynbos would prevent the establishment of forest trees along the margin, long fire-free periods would allow forest expansion into fynbos. Excessive fuel loads (e.g. in fynbos invaded by alien invasive trees and shrubs) may result in fires of extreme intensity and abnormal damage to forest margins (Watson and Cameron 2002).

3.4.5 Fauna

Studies of the effects of fynbos fires on fauna have largely focussed on birds, tortoises, rodents, and rare or endemic species. Faunal abundance or species richness was often considered in relation to fire occurrence, time since fire, or fire season, and the effects of fire on animal-mediated pollination, seed dispersal, seed predation, and grazing systems. Fire effects on other taxa, and those relating to fire frequency, intensity, size, and patchiness are less well understood (Parr and Chown 2003).

Direct mortality due to fire is deemed rare amongst fauna, fire-induced habitat change apparently being more important in determining the fate of animals (van Hensbergen et al. 1992). The geometric tortoise (*Psammodromus geometricus*), a renosterveld endemic, is vulnerable to frequent fires and the destruction of habitat in large fires (Baard 1993), requiring fires at 6–12 yr intervals during January–March to ensure adequate maturation, a supply of herbaceous food plants, and eggs being underground at the time of fire. The parrot-beaked tortoise (*Homopus areolatus*) copes better with frequent fires through higher recruitment rates (Baard 1997). The existence of refuge habitats (such as rock crevices or loose sand for burying) can considerably enhance fire survival in tortoises (e.g. from 14% to 88% in the widespread angulate tortoise (*Chersina angulata*; Stuart and Meakin 1983; Wright 1988)). Strong territoriality, slow colonization rates, and high visibility to predators in burnt environments make some tortoises additionally vulnerable to fires (Wright 1988).

Bird species richness remains largely unchanged following fire, but composition changes and bird abundance increases with time post fire. While some species (e.g. Cape siskin (*Serinus totta*)) are not affected by

vegetation age, non-fynbos specialists and granivores dominate after fire, and nectarivores and some fynbos-endemic birds (i.e. Cape sugarbird (*Promerops cafer*), Protea seedeater (*Crithagra leucopterus*) and Victorin's warbler (*Cryptillas victorini*)) require older (some ≥ 10 yr) vegetation for feeding and breeding (Fraser 1989; Geerts et al. 2012; Lee 2012; Chalmandrier et al. 2013). The bird species turnover that occurs following fire is related to vegetation structure and functional composition, suggesting potentially adverse effects of short FRIs on some species (Martin and Mortimer 1991). Fire seasonality may also affect bird breeding success. Birds in the western inland zone breed largely during July–October (Winterbottom, 1968) suggesting that spring burns will set back breeding bird populations in this zone (van Wilgen and Viviers 1985). Conversely, fynbos fires encourage outbreeding by forcing birds to disperse following the destruction of flowering plants, and this may contribute to high genetic diversity (e.g. in Cape sugarbirds and orange-breasted sunbirds (*Anthobaphes violacea*; Chan et al. 2011)). Limited studies of bird behaviour at fynbos fires or on recent burns (Dean 1989; Fraser 1989) suggest that the relatively low frequency of fynbos fires has not developed the guild of opportunistic, fire-tracking or fire-attending bird species which evolved in savannas.

Pollination rates of ornithophilous plants appear to be negatively affected by fire due to post-fire reductions in bird density (Geerts et al. 2012). Interestingly, the opposite is true for certain insect-pollinated plants, notably oil-secreting orchids, in which pollination rates peak in the first post-fire year and decline sharply thereafter (Pauw 2007). This suggests that different pollination guilds differ in their response to fire (Geerts et al. 2012).

Small mammal abundance is reduced by fire, but populations usually recover within 6–9 months, peaking in the second year post fire (Fraser 1990; Midgley and Clayton 1990; van Hensbergen et al. 1992). There is little evidence for emigration from burnt areas as a result of fires; instead small mammals make use of unburnt refuge habitats (Midgley and Clayton 1990), selecting for dense vegetation cover at or slightly above the ground (van Hensbergen et al. 1992). Fire had little effect on small mammal species composition in the western inland zone (van Hensbergen et al. 1992), while successional patterns with time post fire were evident in the eastern inland and coastal zones, largely determined by vegetation structure (Breytenbach 1987). Species richness appears to increase during the first seven years post fire (Midgley and Clayton 1990; van Hensbergen et al. 1992).

Rodent granivory can reduce seed densities in mature fynbos; it also occurs in young post-fire fynbos but to a lesser extent due to reduced post-fire abundance of rodents (Bond 1984; Bond and Breytenbach 1985; Fraser 1990; Botha and Le Maitre 1992; van Hensbergen et al. 1992). Rodents select seeds based on seed size, hull thickness, and embryo presence, improving their feeding efficiency and determining whether seeds are buried or consumed in situ (Le Maitre 1994; Rusch et al. 2013). Rodents appear to be more important as seed predators in post-fire fynbos than birds and ants (Fraser 1990). Ants reduce predation of seeds by rapidly transporting and burying in their nests both elaiosome-bearing (myrmecochorous) seeds and those without (Bond and Slingsby 1983; Bond and Breytenbach 1985; Pierce and Cowling 1991). In pre- and post-fire environments, seedling mortality due to browsing by rodents varies in severity (Bond 1984; Midgley 1988; van Hensbergen et al. 1992).

While fynbos ecosystems are rarely manipulated specifically for the benefit of indigenous animals, the management of fire to promote grazing for the vulnerable, endemic bontebok (*Damaliscus pygargus pygargus*) in the lowland fynbos of the Bontebok National Park is a notable exception. The effects of the interplay between fire and grazing on large grazing mammals (including the vulnerable Cape mountain zebra (*Equus zebra zebra*)) and the flora have received considerable attention in this park and elsewhere (Watson et al. 2005; Watson and Chadwick 2007; Watson et al. 2011).

3.5 Fire management

3.5.1 Evolution of fire management policies

Fire management policies in the fynbos have changed substantially over the last century, as an understanding of the importance of fires for the conservation and management of healthy fynbos ecosystems grew. These developments have been described in detail elsewhere (van Wilgen, Everson, et al. 1990; van Wilgen et al. 1994, 2011; van Wilgen 2009; Kraaij et al. 2011) and are therefore only briefly outlined here; a summary of key milestones is provided in Table 3.3. European colonists were, for almost three centuries, strongly opposed to the use of fire in ecosystem management and largely practiced fire suppression (Wicht 1945). Prescribed burning was eventually introduced into fynbos protected areas in the 1970s, and was initially carried out during spring under cool, safe conditions. From the 1980s onwards, however, prescribed burning was constrained by ecological restrictions on fire season,

declining funding, safety requirements, and invasive alien plant management considerations. Alternative approaches to fire management were subsequently developed, varying in the degree of manipulation of fire regimes. These include prescribed block burning, systems tolerating wildfires under agreed conditions combined with fuel reduction burning, and 'natural burning zones' where only lightning-ignited fires are allowed to burn. Following the devolution of management responsibilities (mainly to provinces) in 1990, there was an initial sharp decline in management capacity. Some of this capacity has been rebuilt, but it faces significant challenges in the future (van Wilgen et al. 2012).

3.5.2 Challenges and adaptive approaches

The goals of fynbos fire management are largely twofold: (a) to ensure the continued health of the

ecosystem so as to conserve biodiversity and to deliver vital ecosystem services; and (b) to ensure safety and security in a dangerous fire-prone environment. The implementation of effective fire management policies in fynbos ecosystems is bedevilled by a number of challenges, outlined briefly below.

The presence of fire-adapted invasive alien plants

The combination of unplanned wildfires and fire-adapted alien shrubs and trees leads to substantial challenges (see Box 3.1). Planned control operations have to be flexible to accommodate extensive post-fire follow-up if spread is to be contained.

The widespread dominance of unplanned wildfires

Fire managers would ideally like to be able to exercise a greater degree of control on the occurrence of fires, but are for all purposes prevented from doing so

Box 3.1 Fire and invasive alien plants in fynbos

Alien plants that become invasive in fynbos are of necessity fire-adapted. The most problematic species are trees and shrubs of two types. The first are serotinous pines (*Pinus* spp.) and hakeas (*Hakea* spp.) which are spread mainly by mass seed release following fire. The seeds of both *Pinus* and *Hakea* are winged, and can disperse over considerable distances post fire, assisting their establishment in remote areas, where access for control is difficult. The second type includes trees and shrubs in the genus *Acacia*, which produce copious amounts of hard-coated, long-lived, soil-stored seeds. These seeds are stimulated to germinate by fires, resulting in dense impenetrable stands after a few fire cycles.

Besides their impacts on biodiversity and water resources (van Wilgen et al. 2008), invasive alien plants can increase above-ground biomass, thus adding to fuel loads and increasing fire intensity under certain conditions, and leading to undesirable consequences such as soil damage, erosion (Scott et al. 1998), and increased difficulty of fire control. Invasive alien plants arguably pose the greatest threat to the long-term survival of fynbos ecosystems. There is therefore an imperative to control them, in conjunction with fire where necessary.

Pines and hakeas are relatively easy to kill (they can be felled and do not resprout) and their seeds are released from serotinous cones or follicles after death of the parent plant. These seeds are less prone to be spread by wind (because the parent plants are prone, not upright) and either are con-

sumed by rodents or germinate. Resultant seedlings can be killed by following felling with a prescribed fire a year later, and this can lead to their virtual elimination. Despite this, control is difficult. Unplanned wildfires are commonly responsible for ongoing seed release and spread to inaccessible areas, where the invasive populations cannot be reached. Biological control can reduce seed production in hakeas, leading to a degree of control (Esler et al. 2010), but such solutions are not available for pines, which have commercial importance in the CFR (Hoffmann et al. 2011).

Australian acacias are more difficult to control. These species are not killed by felling, and application of herbicides to cut stumps is required to prevent resprouting. Such treatments are usually followed by burning to stimulate germination from the soil seedbank, but the resultant flush of seedlings demands exhaustive follow-up if the desired reduction in density of invasive plants is to be achieved. Fortunately, biological control of Australian acacias has shown promise in reducing both the vigour of the plants and the levels of seed production, making control more feasible (Moran and Hoffmann 2012).

Threats posed to fynbos by alien invasive (annual and perennial) grasses have received comparatively little attention, but fires promote their recruitment which impedes recovery of lowland fynbos after fire or other disturbances (Musil et al. 2005; Holmes 2008). These grasses furthermore have the potential to alter fire regimes (Parker-Allie et al. 2009), typically increasing fire frequency.

Table 3.3 Milestones in the development of a fire management policy for fynbos ecosystems.

Date	Event	Significance for fire management
1924	Royal Society condemns veld burning	Ban on all fires
1945	Publication of a report on the preservation of vegetation in the southwestern Cape	First serious suggestion of prescribed burning
1946	Soil Conservation Act passed by Parliament	Formation of Fire Protection Committees to suppress wildfires
1949	Documentation of the youth periods for <i>Protea repens</i>	Understanding of minimum interval between fires
1962	Wildfire sweeps through senescent stand of <i>Serruria florida</i> ; profuse regeneration occurs	Realization that species need fire
1968	<i>Orothamnus zeyheri</i> populations senesce and die; decision to burn sites to stimulate germination	First official prescribed burn in fynbos
1970	Mountain Catchment Act passed by Parliament	Increase in the area under management by prescribed burning
1970	Fire frequency of eight years suggested for Marloth Nature Reserve near Swellendam	First guidelines for ongoing implementation of prescribed burning
1974	Forestry Department expands research programme	Increase in understanding of the role of fire
1978	Initiation of the Fynbos Biome Project	Increase in understanding of fynbos ecology
1979	Proposal of broad rule that would justify burning when obligate reseeding plants had flowered for three successive seasons	Refinements to guidelines for ongoing implementation of prescribed burning
1980	Introduction of natural fire zone management in the Swartberg	Departure from rigid approach to prescribed burning
1984	Publication of papers demonstrating negative effects of winter and spring burns on obligate reseeding Proteaceae	Seasonal restrictions on prescribed burning; burning plans fall behind schedule
1990	Responsibility for management of most fynbos areas devolved from national government to provinces; South African Forestry Research Institute moved to the Council for Scientific and Industrial Research (CSIR)	Loss of management and research capacity
1998	Veld and Forest Fire Act passed by Parliament	Explicit recognition that ecological requirements be considered along with fire hazards in fire management; establishment of Fire Protection Associations with mandates for integrated fire management
2003	Launch of the Working on Fire programme	Increased capacity for integrated fire management
2001 and 2011	Publication of proposals for adaptive approach to fire management in fynbos protected areas	Management decisions guided by multiple considerations, taking wildfires and the need for variability into account

Sources: van Wilgen, Everson, et al. 1990; van Wilgen et al. 1994, 2011; van Wilgen 2009; Kraaij et al. 2011.

by wildfires which currently account for almost 90% of the total area burnt. Funds for fire management are correspondingly and overwhelmingly devoted to wildfire suppression, leaving little scope to conduct prescribed burns.

The conflict between ecological and safety requirements

The Veld and Forest Fire Act caters for integrated fire management, recognizing both the ecological role of fire for maintaining healthy ecosystems, and the need to reduce the risks posed by fires. In reality, however, safety considerations often take precedence over ecological considerations (van Wilgen and Richardson

1985; Kraaij et al. 2011; van Wilgen et al. 2012). In particular, the need for a substantial proportion of relatively high-intensity fires can create conflict (van Wilgen et al. 2011), especially because the window of opportunity to conduct such burns during the dry warm season in the western and inland zones is very small. A lack of restraint on fire season in the far eastern coastal zone (Kraaij 2012) gives fire managers more latitude although the need for high fire intensity remains.

Altered patterns of ignition and fire spread

Landscape fragmentation and transformation, especially in lowland fynbos, result in fires no longer reaching isolated patches, demanding a level of fine-scale

management that is seldom practical or affordable. In addition, the number of ignitions grows as human populations grow, increasing the frequency of fires and threatening biodiversity.

Global climate change

Predictions are that the incidence of weather conditions conducive to the initiation and spread of fires will increase as the global climate changes. This will bring further pressures to bear on fire managers.

These challenges add a large degree of uncertainty and complexity to fire management, and the only effective way to deal with this would be to adopt an adaptive approach to management. New proposals for adaptive fire management in fynbos ecosystems (van Wilgen et al. 2011; Kraaij, Cowling, and van Wilgen 2013b) have followed the development of similar approaches in savanna areas (van Wilgen and Biggs 2011). Management should be based on the acceptance of an agreed vision, desired state, and objectives; the formulation of management targets (thresholds of potential concern) which describe the boundaries of the desired state; and the implementation of monitoring to assess whether thresholds are being approached or exceeded. If they are, management actions could be formulated to address this.

Proposed 'operational' thresholds guide managers with respect to achieving target fire patterns and include, for example, thresholds relating to the proportional area occupied by different post-fire age classes, or the proportion of area burnt at different FRIs over the past few decades. Each age class, or FRI class, is assigned upper and lower thresholds (e.g. the proportion of area in each age class should be >5% but <20%). Exceeding these thresholds would trigger management action to bring the system back within thresholds. 'Ecological' thresholds are based on attributes of selected indicator species (e.g. proportion of populations that have flowered for three or more seasons, proportions showing signs of senescence, or trends in population size). Exceeding an ecological threshold would indicate an undesirable condition that could be addressed through the appropriate application or withholding of fire. Management would be adaptive because actions would be guided by new insights gained through monitoring and assessment. The system holds promise, but is still in its infancy in fynbos fire management (van Wilgen et al. 2011). The thresholds briefly outlined here have only recently been proposed by researchers, and need to be formally accepted and incorporated into management plans. This will almost certainly come about, since a return

to a system of extensive prescribed burning, which is ineffective for a number of reasons, is clearly not an option.

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