# Fire management in species-rich Cape fynbos shrublands

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**The management of fire-dependent biodiversity hotspots must be based on sound ecological knowledge and a pragmatic approach that accommodates the constraints within which fire managers must operate. South Africa's fynbos biome (shrubland or heathland vegetation found in the Western Cape of South Africa) is one such hotspot. In this region, the implementation of prescribed burning to conserve biodiversity must take into account the area's rugged and inaccessible terrain and recurrent wildfires, the presence of fire-adapted invasive alien plants, and the imperatives for ensuring human safety. These constraints limit the potential for prescribed burning to be effective everywhere, and prioritization and trade-offs will be needed to ensure the efficient use of limited funding and management capacity. In such environments, management must be adaptive, based on clearly defined and shared goals, monitoring, and assessment, and should be flexible enough to adjust as new lessons are learned.**

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There is growing appreciation of the importance of the Earth's biodiversity and the need to conserve as much of it as possible (Cardinale *et al.* 2012; Reich *et al.* 2012). In fire-prone and fire-adapted ecosystems that are also rich in species, fires are needed to maintain high levels of biodiversity, and this in turn requires careful management. South Africa's fynbos biome, located in the country's Western Cape and Eastern Cape Provinces, is dominated by fire-prone, Mediterranean-climate shrublands. The biome was included as one of 25 biodiversity hotspots that together comprise 1.4% of the land surface of the world, and that should be regarded as conservation priorities on the basis of exceptional concentrations of endemic species (Myers *et al.* 2000). The fynbos biome is also recognized as one of six floristic kingdoms in the world (Takhtajan 1986), and is the only one that is wholly within the borders of a single country. Eight fynbos protected areas were recently proclaimed as World Heritage Sites by the United Nations, in recognition of their global importance as centers of endemism (Figure 1). Detailed plans have been developed to guide conservation actions in the region (Cowling *et al.* 2003), while the main focus over the past decade has been on establishing protected areas and improving capacity for their management (Gelderblom *et al.* 2003).

Prescribed burning has been regarded as an important management practice in fynbos ecosystems for over 40 years (van Wilgen 2009). While indigenous African peoples have both tolerated and used fire for millennia, European colonists were, for almost three centuries, strongly opposed to its use in ecosystem management. It was only in the mid-20th century that Wicht (1945) suggested that the phenomenon of fire deserved closer scrutiny, to assess both its role and potential use in fynbos management. Prescribed burning was eventually introduced into fynbos protected areas in the 1970s, at which time its major goals were to ensure a sustained flow of water from adjacent mountain areas and for nature conservation (Bands 1977). Subordinate goals included the control of invasive alien plants and fuel reduction to protect plantations of alien pines (Bands 1977).

In this paper, I review the various approaches to fire management that have evolved in fynbos ecosystems over the past half century and examine the challenges associated with the use of fire for maintaining species diversity in an exceptionally species-rich ecosystem.

### In a nutshell:

- South African fynbos shrublands are a biodiversity hotspot of global importance
- Fynbos vegetation is fire-prone and fire-adapted, and managing fynbos equates to managing fires
- Prescribed burning in the fynbos has been promoted for over 40 years but has only accounted for ~11% of the area burned because of numerous constraints
- There is a clear need to develop a better understanding of how the limited opportunities for prescribed burning can be strategically used to maximize the benefits for conserving the region's remarkable biodiversity through trade-offs and adaptive learning

## The fire ecology of Cape shrublands

Fire is the principal driving force in the ecological dynamics of fynbos ecosystems, and fires are both inevitable and necessary (Panel 1; Figure 2). Managing fynbos equates to managing fire, and a thorough understanding of fire regimes (the combination of fire fre-

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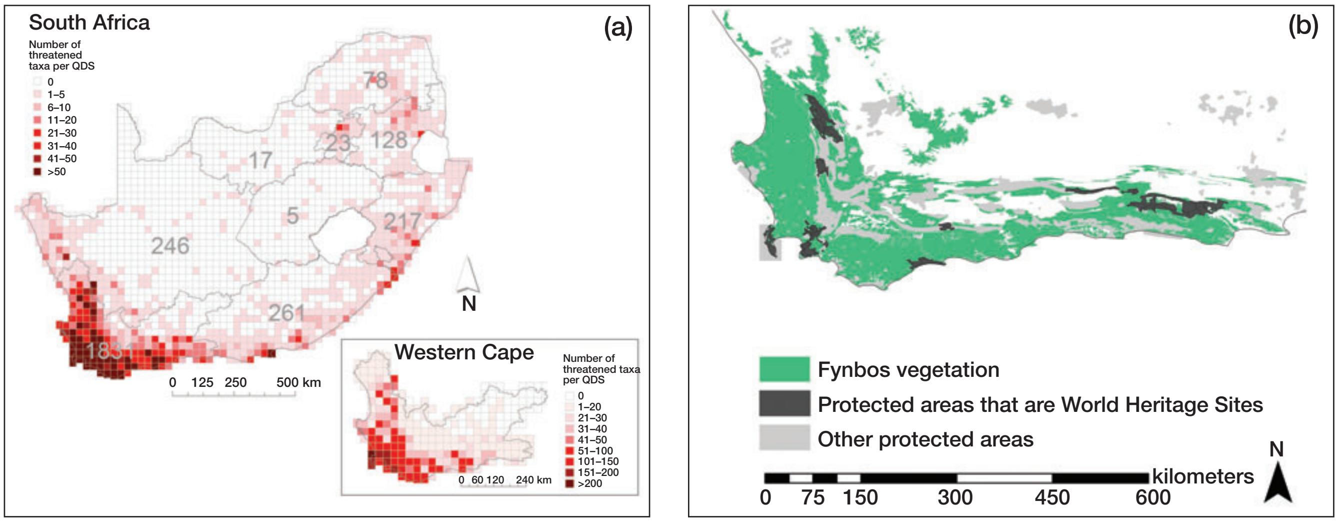


***Figure 1.*** *(a) The number of threatened plant taxa (critically endangered, endangered, and vulnerable) per quarter degree square (QDS, approximately 25 x 25 km) in South Africa, showing their concentration in the fynbos biome. Numbers show totals per province; 65% of the country's threatened taxa are concentrated in the Western Cape Province (inset). The fynbos biome contains over 9000 plant species (6210 endemics); 3087 plant taxa (2972 endemics) are of conservation concern and 1736 taxa (1690 endemics) are in danger of extinction (maps and data from Raimondo* et al. *2009). (b) The fynbos biome covers 83 946 km$^2$; 68.4% remains untransformed and 10.1% lies within protected areas.*

quency, season, and intensity that characterize a region; Gill 1975) and ecosystem responses to fire are prerequisites for effective management. Fires occur in fynbos at intervals of 10–15 years across the biome (van Wilgen *et al.* 2010; Kraaij *et al.* 2011). Repeated short (4–5 years) intervals between fires can eliminate dominant reseeding shrubs, and increased fire frequency favors resprouting species. Decreases in plant diversity have been noted where resprouting species become dominant (Vlok and Yeaton 1999). Fires are strongly seasonal and concentrated in summer in the west of the biome, but are markedly less seasonal in the east, where rainfall is evenly distributed across all seasons (Figure 3). Fire season affects the germination and survival of seedlings of dominant shrubs, and these effects are therefore noticeable in the west (Bond *et al.* 1984) and almost absent in the east (Heelemann *et al.* 2008; Kraaij *et al.* 2013c). Typical fynbos fire regimes are also characterized by variability around the mean fire return interval and season (Forsyth and van Wilgen 2008; van Wilgen *et al.* 2010), and this variability may well be necessary for plant species coexistence (Cowling and Gxaba 1990; Thuiller *et al.* 2007). Fire return intervals, fire season, and fire intensity are almost never the same between successive fires at the same location. As a result, the density of overstory shrubs changes between successive fires, and this variation in overstory shrub density is in turn associated with the maintenance of diversity in understory species (Cowling and Gxaba 1990; Thuiller *et al.* 2007). Pre-fire shrub densities were also found to affect the density of post-fire recruitment (Bond *et al.* 1995), resulting in alternating densities and species diversity at the same site between successive fires. Thuiller *et al.* (2007) concluded that recurrent fires buffer plant populations from the threat of extinction by ensuring stable coexistence over time, despite localized extirpations caused by individual fires. There is growing concern, however, that repeated short intervals between fires are becoming too common (Forsyth and van Wilgen 2008), which may negatively affect populations of obligate reseeding plants that would not have sufficient time to mature and set seed between fires (van Wilgen and Forsyth 1992). While variation in fire regimes may be acceptable and even necessary, there are limits beyond which elements of the vegetation may be negatively affected. From a conservation perspective, a pattern of repeated, widespread short-interval fires will almost certainly be undesirable.

Fynbos shrublands (which grow primarily on nutrient-poor sandstone substrates) are not the only vegetation type in the biome. Other dominant types include renosterveld shrublands, which grow on nutrient-rich shale substrates (Panel 2; Figure 4), strandveld shrublands, found on coastal sands and limestone substrates, and Afromontane forests that are confined to sheltered ravines or fire-free areas (Mucina and Rutherford 2006). While there is a reasonably good understanding of fire ecology in fynbos, the same cannot be said for other vegetation types found in the biome. Fire-prone renosterveld shrublands are poorly understood and their fire ecology has not been well studied (Keeley *et al.* 2012). Strandveld shrublands do occasionally burn, but they are not particularly fire-prone and little is known about the effects of fire on these coastal ecosystems; they are dominated by broad-leaved species, and fire-related recruitment traits and fire-stimulated flowering are absent (Keeley *et al.* 2012). Afromontane forests occur in sheltered ravines, and fires do not penetrate these forests because the potential fuel occurs in compacted, relatively moist litter layers that do not burn (van Wilgen *et al.* 1990); in the eastern sections of the biome, larger Afromontane forest patches persist where the topography shelters them from hot, dry winds that drive fires (Geldenhuys 1994). Many areas occupied by fynbos would become forest in the absence of fire (Manders and Richardson 1992).

A broad understanding of fire ecology has informed the development of general principles to guide prescribed burning (van Wilgen *et al.* 1992), but several issues remain contentious. These include whether or not burning can have detectable effects on fire regimes measured over longer periods, and what the implications are of shifts in fire regimes for the conservation of biodiversity. In particular, there are concerns that prescribed burning would not be appropriate in an environment where human populations increase the risks of ignition and fires are becoming more frequent (van Wilgen *et al.* 2010). The co-occurrence of forest patches in fire-prone shrublands is a unique feature of the fynbos biome. Early approaches to management sought to protect these forests from fire, but this led to their expansion at the expense of fynbos and threatened to eliminate some endemic and fire-dependent fynbos species (van Wilgen *et al.* 2012a). However, the desired balance between fire-dependent fynbos and fire-sensitive forests is seldom specified in fire management plans. Finally, the widespread establishment of fire-adapted invasive alien plant species in the fynbos biome presents substantial challenges to ecosystem managers.

## Prescribed burning in fynbos ecosystems

The introduction of prescribed burning in fynbos ecosystems in the 1970s replaced a policy of fire suppression. The new policy recognized that fires were necessary and that by burning under safer conditions, the incidence of damaging wildfires could be reduced (van Wilgen 1984). Initially, prescribed burning was conducted in fixed areas ("blocks"; van Wilgen 2009), which allowed managers to schedule fires with the goal of imposing a desired fire regime. Prescribed block burning was widely implemented every spring between 1970 and 1980, when relatively cool, moist and wind-free weather conditions allowed for safer burning. However, a steady decline in prescribed burns followed policy directives that restricted burning to late summer or early autumn (van Wilgen 2009). This arose from research suggesting that burning in spring would have detrimental effects on the vegetation (Bond *et al.* 1984), thus eliminating the possibility of

**Panel 1. Fynbos fire ecology**

Most extant fynbos vegetation occurs in rugged mountain areas, which makes access difficult and complicates fire management (Figure 2a). Dry-season fires are a common feature of the fynbos biome, and wildfires account for almost 90% of area burned, despite policies that promote prescribed burning (Figure 2b). Dominant *Protea* shrubs are killed in fires, and seeds (light brown patches below dead shrubs) are released en masse from serotinous flower heads to germinate following the onset of cold weather (Figure 2c). Most plant species resprout post-fire, and mass flowering often occurs in geophytes (plants with underground storage organs) in the first spring following fire (Figure 2d). Fynbos vegetation is regarded as mature (shrubs retaining serotinous flower heads from several past years) at post-fire ages between 10–30 years, at which age seed reserves are sufficient to replace parent populations killed by fire (Figure 2e). If the vegetation exceeds this age, senescence can set in (Bond 1980; van Wilgen 1982) and seed reserves become depleted. Fire-free forest patches are embedded in fynbos shrublands in sheltered ravines (Figure 2f).

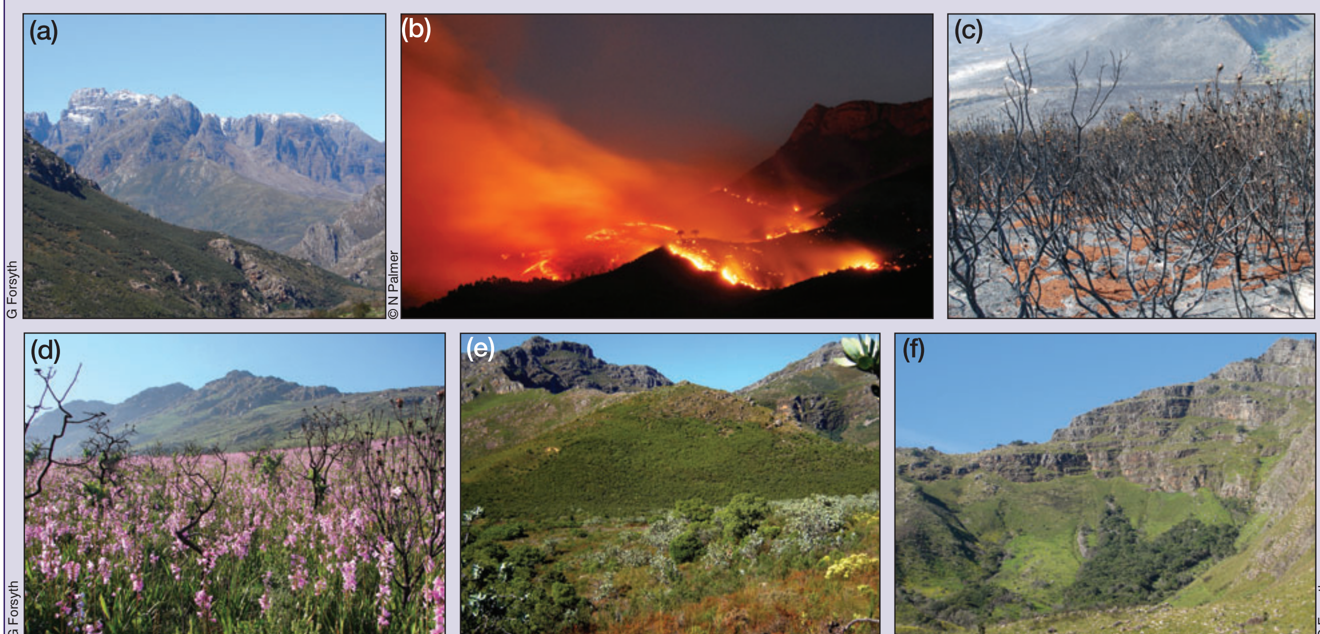


***Figure 2.** Examples of fynbos ecosystems and vegetation at different post-fire ages.*

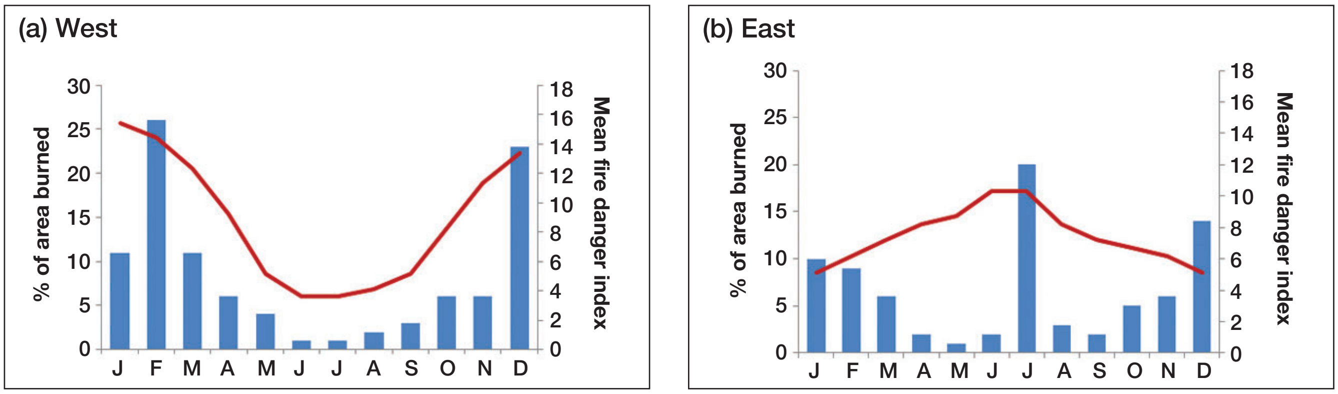


***Figure 3.*** *Mean monthly fire danger indices (red lines; McArthur's Forest Fire Danger Index; Noble* et al. *1980) for typical western and eastern locations in the fynbos biome, and percentage total area burned (blue bars) from selected local fire records. Data are from (a) the Cedarberg and (b) the Outeniqua mountain ranges (van Wilgen and Burgan 1984; van Wilgen* et al. *2010; Kraaij* et al. *2013a, 2013b).*

burning in the austral spring months of September and October, the months with the best weather for burning (van Wilgen and Richardson 1985). As a result, the area burned had declined by 75% by 1988, and the original plan to subject all areas to prescribed burns on a 12-year cycle fell behind schedule (van Wilgen *et al.* 1990). Other factors related to funding, safety, and invasive alien plant management also constrained the application of prescribed burning in the late 1980s (van Wilgen 2009). All of these difficulties led to the development of alternative approaches to fire management (Seydack 1992; van Wilgen *et al.* 1994). These varied in the degree of manipulation of fire regimes and included several options. Some areas were earmarked for management by prescribed burning, where the intention was to burn as much of the area as possible through managed fires. Other systems tolerated wildfires under agreed conditions, and depended on opportunities created by fuel reduction in wildfires to safely burn other areas. Finally, "natural burning zones" were designated, where no prescribed burning was done and natural (lightning-ignited) fires were allowed to burn. However, despite a 40-year policy that advocated prescribed burning in fynbos, prescribed burns only accounted for 11.3% of the area burned between 1970 and 2007 (van Wilgen *et al.* 2010).

## The influence of prescribed burning on fire regimes

The ultimate goal of fire management for biodiversity conservation is to ensure the perpetuation of a fire regime to which the biota are adapted and under which they will thrive. The question of whether or not prescribed burning influences fire regimes is therefore an important one. A few studies have indicated that fire management has had little effect on fire regimes. The replacement of a policy of fire suppression with one of prescribed burning in fynbos ecosystems in 1970 was aimed at reducing the number and extent of wildfires; however, one study found that the area burned in wildfires actually increased following this change in management approach (Brown *et al.* 1991). Another study showed that successive policies that included (1) burning to facilitate grazing, (2) fire suppression, and (3) the promotion of a natural (lightning-driven) fire regime over 70 years had little effect on fire regimes; the extent of burning followed climatic cycles, and fire patterns were "largely unaffected by the

**Panel 2. Fire management in ecosystem fragments**

Renosterveld shrublands were the dominant vegetation on the relatively nutrient-rich, shale-derived soils in the lowlands of the fynbos biome. These shrublands have largely been converted to crop agriculture (mainly wheat and canola), and less than 4% now survives as remnant fragments (Figure 4). Both fire and grazing by large mammals would have been important disturbances in these ecosystems, but their role has been poorly studied. Large numbers of threatened and endemic plant species occur in these fragments, and their conservation depends to some degree on maintaining appropriate fire regimes in these patches. Their small size, scattered distribution, occurrence largely on privately-owned land, and the poor understanding of the role of fire in renosterveld shrublands hinders the implementation of a coherent program of prescribed burning.

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***Figure 4.*** *A remnant of renosterveld vegetation surrounded by cultivated fields of canola.*

 

absence or presence of fire control measures" (Seydack *et al.* 2007). Almost all fynbos areas have sufficient fuel to support the spread of fires at a relatively young (5 years) post-fire age, but inland zones experience a more severe fire climate than do coastal zones (~35% versus 15% of days with high to very high fire danger, respectively; van Wilgen *et al.* 2010). Despite these differences, fire return periods were similar at inland and coastal locations (10–13 years). Van Wilgen *et al.* (2010) concluded that the frequency of fires at inland locations was limited by fewer ignition opportunities that offset the more severe weather conditions. Higher human population densities in coastal locations increased the chances of igniting accidental fires despite milder weather. It appears that fynbos fire regimes remain largely unaffected by prescribed burning and are driven by mostly unplanned ignitions.

## ■ Fire and alien plant invasions

Invasion by fire-adapted non-native trees and shrubs is arguably the biggest threat to fynbos ecosystems (Panel 3; Figure 5). Introduced Australian trees and shrubs in the genus *Acacia* produce an abundance of seeds that build up in the soil; soil and seeds are spread by humans when soil is transported, and are also carried downstream when rivers are in spate. This leads to new foci of *Acacia* invasion along roads (where downhill spread occurs) and increased establishment along rivers and streams. Seeds are stimulated to germinate en masse by fire, so burning can dramatically increase the amount of new growth under parent trees and along rivers (Figure 5a). Felling followed by burning has been used to deplete soil-stored seed banks but could not be applied over large areas due to the need for repeated and intensive follow-up weeding of emergent seedlings (Pieterse and Cairns 1986).

The introduction of biological control to reduce the seed output of *Acacia* species has substantially changed this picture in recent years. A suite of seed-feeding weevils and gall-forming flies and wasps (which prevent seed production by inducing the formation of galls instead of seed pods), have substantially reduced the seed output of many *Acacia* species (Figure 5b). This in turn has increased the prospects for effective control through the combination of mechanical felling, fire, and seed reduction (Moran and Hoffmann 2011).

Serotinous trees and shrubs in the genera *Pinus* (pine trees from Europe and North America) and *Hakea* (proteaceous shrubs from Australia) are killed by fires but spread over considerable distances by means of winged seeds that germinate in the post-fire environment. Control is possible through prefire felling (after which seeds are released) and burning after 1–2 years (which kills any resultant seedlings). The control is effective, but because of the difficulties of reaching stands in remote and rugged terrain, success in mechanical clearing of invasive *Pinus* and *Hakea* plants has been limited (van Wilgen *et al.* 2011a). In the case of *Hakea*, biological control in the form of a seed-feeding weevil and a seed-feeding moth has been in place for several decades, which has led to some progress in the control of *Hakea* species (Esler *et al.* 2011). There are no biological control agents approved for *Pinus* species, however, due to concerns about the impact they may have on the region's forest industry (Hoffmann *et al.* 2011). Plantation-based forestry using invasive alien pine trees provides economic benefits to local human communities, but also supplies a ready source of seeds to invade adjacent fynbos areas. The net value of plantations could well be negative if the impacts of invasion, and of increasing destruction of plantations by wildfires, are factored in. This has led to suggestions that pine-based forestry in the fynbos biome should be phased out (van Wilgen and Richardson 2012), but such proposals are understandably controversial.

**Panel 3. Invasive alien plants and fire**

Fire-adapted invasive alien plants are arguably the largest threat to the integrity of fynbos ecosystems. Invasive Australian *Acacia mearnsii* trees dominate along a river (Figure 5a); their control requires felling and burning to stimulate seed germination and deplete soil-stored seed pools. The prospects for control have been substantially increased through the introduction of biological control agents that lay eggs in developing flowers so that they form galls instead of seed pods, preventing seed production (Figure 5b). Fire-adapted pine trees are spread by winged seeds and have to be manually felled and then burned to destroy seedlings (Figure 5c).


***Figure 5.*** *Invasive alien trees in fynbos ecosystems.*

Finding a sustainable solution to the fire-driven, non-native pine invasion problem is possibly the most important challenge facing managers of fynbos ecosystems.

## ■ Managing dual goals

The reconciliation of fire management goals that relate to safety on the one hand and to the maintenance of ecosystem health on the other remain among the most important and controversial aspects of fire management (van Wilgen *et al*. 2012a). Legislation governing the management of bushfires in South Africa is centered on 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, those responsible for the implementation of the legislation adopt a primary focus on risk reduction and safety at the expense of ecological considerations. Ecosystem health is best maintained by promoting a variable fire regime that will benefit all elements of the biota, and that must include a substantial proportion of relatively high-intensity fires in the dry summer season. In contrast, ensuring safety requires the prevention and suppression of high-intensity dry-season fires that threaten human lives and infrastructure. The safe achievement of ecosystem health goals requires prescribed burning under milder weather conditions; the opportunities for conducting prescribed burns are constrained by the relatively few suitable days in early autumn, when weather conditions coincide with the ecologically acceptable burning season (van Wilgen and Richardson 1985); this has been a primary reason why sufficient and effective prescribed burns have not been carried out (Panel 4; Figure 6).

## ■ Burning for animal conservation

While fynbos ecosystems are rarely manipulated specifically for the benefit of indigenous animals, there are some notable examples involving rare endemic species that have influenced fynbos fire management. One of Africa's rarest antelopes, the IUCN Red Listed "Vulnerable" bontebok (*Damaliscus pygargus pygargus*; Figure 7a), historically inhabited the nutrient-rich renosterveld shrublands that have now largely been converted to crop agriculture (Figure 4); this has restricted the remaining bontebok populations to suboptimal, nutrient-poor habitat in small protected areas. Beginning in the early 1970s, short-interval burning was implemented in Bontebok National Park in Western Cape Province to promote grazing. As a result, post-fire age distribution was highly skewed toward young vegetation, with detrimental effects on slower maturing shrubs (Kraaij 2010). From 2004 onwards, the fire interval was lengthened to favor plant species diversity, an increasingly urgent conservation priority for the park. Managers of these protected areas have recognized that the objectives of simultaneously conserving large grazing mammals and maintaining plant species diversity may be difficult to reconcile in the highly fragmented renosterveld and lowland fynbos ecosystems (Novellie and Kraaij 2010). As a result, the abundance of large grazing mammals has been reduced (including some bontebok that have been translocated to other protected areas), and plant species have been selected as indicators to monitor the effects of grazing (Novellie and Kraaij 2010).

The endemic Cape sugarbird (*Promerops cafer*; Figure 7b) is an important pollinator of dominant proteaceous shrubs in fynbos areas. Sugarbirds frequent mature (8–20 year post-fire) vegetation, and are largely absent from more recently-burned areas. Concerns have been

**Panel 4. *Mimetes stokoei* and the failure of prescribed burning**

*Mimetes stokoei* (Proteaceae; Figure 6) is a spectacular flowering shrub that is relatively short-lived and survives interfire periods as seeds that are buried by ants (myrmechochory; Bond and Slingsby 1983). Myrmecochorous seeds require fires for optimal germination, both to scarify the hard seedcases and to provide optimal soil temperature regimes for germination (Brits 1986). *M stokoei*, never common, survived as a single seedling in 1967, but this subsequently died. The area of known last occurrence was burned in a prescribed fire in 1971, and a further area was burned in a "slow, cool" prescribed fire in 1984 (Slingsby and Johns 2009), but no seedlings emerged from either burn and the species was declared extinct shortly afterward (Hilton-Tailor 1996). The site was burned again in a wildfire in 1999 ("an uncontrollable blaze in rising hot, dry winds"; Slingsby and Johns 2009) and 24 seedlings emerged. The species' status was revised to "critically endangered" following this discovery (Raimondo *et al.* 2009). This illustrates the failure of "safe" prescribed burning to achieve conservation objectives – greater fire intensity was needed to stimulate *M stokoei* germination, as demonstrated for other *Mimetes* species (Bond *et al*. 1990; van Wilgen *et al.* 1992). However, higher intensity fires would in all likelihood not be permitted for safety reasons. This case epitomizes the dilemma faced by ecosystem managers who are simultaneously required to promote ecosystem goals while ensuring human safety.


***Figure 6.*** Mimetes stokoei *shrubs at 5-years post-fire age.*

 




***Figure 7.*** *Endemic fynbos animals. (a) Bontebok were once common on renosterveld shrublands but are now confined to patches of marginal habitat due to widespread conversion of suitable habitat to agriculture; (b) Cape sugarbirds require mature vegetation in older post-fire age classes and are important pollinators of* Protea *shrubs; (c) geometric tortoises are also confined to a few remnant patches of renosterveld shrublands, and are vulnerable to frequent and large fires.*

expressed that increases in fire frequency will lead to declining populations of this species, with cascading effects on plant species that depend on sugarbirds for pollination (Geerts *et al.* 2012). Fire managers must therefore attempt to retain enough vegetation in the older post-fire age classes to prevent declines in sugarbird populations. Until recently, this had not been recognized as a serious management consideration.

The geometric tortoise (*Psammobates geometricus*; Figure 7c) is an endemic reptile confined to a few small remnants of renosterveld shrubland. Young tortoises are vulnerable to frequent fires, and large fires can remove suitable habitat (Baard 1993). Because the remaining habitat is largely fragmented and occurs in relatively small patches, fire managers should seek to create a mosaic of different post-fire vegetation ages; this would reduce the impact of large fires and ensure sufficient tortoise habitat at any given time (Baard 1993). This has not, however, been implemented in practice, partly because much of the remaining suitable land is under private ownership. Large wildfires continue to place geometric tortoise populations under stress and they face an uncertain future unless a concerted effort can be made to implement effective fire management.

## ■ Adaptive management

Ecosystem responses to management are complex, making the outcomes of management interventions difficult to predict (Keith *et al.* 2011). The limited use of prescribed burning, the lack of understanding of how most species respond to fire, and the need to incorporate socioeconomic considerations into fire management have led to calls for the adoption of adaptive approaches over the past two decades. Guidelines for deciding when and where to burn fynbos were developed in the 1980s, based on the known responses of a few fynbos species to variations in fire frequency and season (van Wilgen *et al.* 1992). These species were mainly obligate reseeding proteaceous shrubs, but almost nothing is known about the fire ecology of the five largest plant genera in the fynbos: *Erica* (658 species), *Aspalathus* (272 species), *Pelargonium* (148 species), *Agathosma* (143 species), and *Phylica* (133 species). It was realized early on that the widespread use of guidelines based on relatively few species may favor certain species while disadvantaging others, and that varying the fire regime may be desirable for the long-term conservation of the full suite of species (van Wilgen *et al.* 1994). However, monitoring programs designed to track the impact of burning on shrub persistence failed to produce any clear links between burning and ecological outcomes, and fell into disuse (van Wilgen *et al.* 1994). In addition, there was a need to accommodate inevitable wildfires into flexible planning approaches, especially as these wildfires disrupted plans for invasive alien plant control. To accommodate these needs, a computer-based fire management system was designed to monitor fire patterns as they established, and to prioritize areas for the strategic application of fires that would ensure a variety of post-fire ages in the landscape (Richardson *et al.* 1994). Although the system was never used in practice, the potential for geographic information system (GIS) technology to support management decisions has grown substantially over the past few years. GIS is now routinely used to analyze fire patterns over time (Seydack *et al.* 2007; Forsyth and van Wilgen 2008; van Wilgen *et al.* 2010; Kraaij 2010; Kraaij *et al.* 2013b), but the need to accommodate non-ecological considerations (such as safety when burning) and differences in the risk appetite of fire managers continue to be a challenge with respect to the application of fire (van Wilgen *et al.* 2012a).

New proposals for adaptive fire management in fynbos ecosystems (van Wilgen *et al.* 2011b; Kraaij *et al.* 2013c) have followed the development of similar approaches in savanna areas (van Wilgen and Biggs 2011). Management should be based on a vision agreed upon by all stakeholders, the resultant desired state, and objectives; the formulation of management targets (thresholds of potential concern) that describe the boundaries of the desired state; and the implementation of monitoring programs to assess whether thresholds are being met or exceeded, and for which management actions could be formulated. Proposed "operational" thresholds guide managers with respect to achieving target fire patterns and include, for example, the proportions of burned area in different post-fire age classes or the proportion of area burned at different fire return intervals over the past few decades. Each age class, or fire return interval class, is assigned upper and lower thresholds (eg 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 threshold parameters. "Ecological" thresholds are based on attributes of selected indicator species (eg proportion of populations that have flowered for three or more seasons, proportions showing signs of senescence, 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.* 2011b). The thresholds outlined briefly here have only recently been proposed by researchers, and need to be formally accepted and incorporated into management plans. These systems of adaptive management would allow more informed decisions to be made regarding where and when to use fire. Their adoption seems to be the best option for the future, as the alternatives (regular, scheduled prescribed burning or wildfire suppression) do not promote the acheivement of management goals.

## ■ Fynbos in the context of other Mediterranean-climate ecosystems

Fire management in fynbos ecosystems bears strong similarities to other areas in fire-prone Mediterranean climates, but there are also important differences. The key difference between fynbos and comparable systems elsewhere is that relatively low-nutrient environments in the Cape (and Australia as well) have driven the evolution of vegetation that is much richer in species diversity than that of Northern Hemisphere areas (Keeley *et al.* 2012), and the need to tailor fire regimes more tightly to biodiversity outcomes is therefore higher in the Cape than elsewhere. Other differences arise both from ecological and social features of the particular regions. In Californian chaparral and Mediterranean-basin ecosystems, major wildfires encroach on the urban environment, destroying homes and causing deaths on a regular basis. For example, recent fires in California, Portugal, and Greece destroyed thousands of buildings, with substantial loss of human life (Keeley *et al.* 2012). In contrast, Western Australia and South Africa do not experience such damage, despite the frequency of fires (McCaw and Hanstrum 2003; Forsyth and van Wilgen 2008). Even the most destructive fires on record in fynbos have destroyed very few houses, and loss of life is rare (Forsyth and van Wilgen 2008). The reasons for these differences include relatively low human population densities and a "hard" urban edge that separates natural vegetation from developed areas, less severe fire weather, and lower fuel loads and more frequent fires that do not allow for extensive build-ups of fuel (Keeley *et al.* 2012). Nonetheless, the damage that is occasionally inflicted by wildfires has been sufficient to make prescribed burning subject to strict control, especially when close to developed areas, making it difficult to carry out (van Wilgen *et al.* 2012a). In California, fire frequency has increased over the past century, linked to growing human populations and increasing sources of ignition (Keeley and Fotheringham 2001); a similar problem now appears to be manifesting itself in some Cape fynbos areas (Forsyth and van Wilgen 2008; van Wilgen *et al.* 2010).

## ■ Conclusions

Fynbos ecosystems are highly diverse and worthy of conservation efforts. Fire management must form an important component of such efforts, and ecologists have provided a relatively sound foundation of knowledge on which to build. The dynamics of fynbos ecosystems in relation to fire are relatively well understood, conservation planning has identified priority areas for protection, and strong guidelines are available for managing fires to achieve ecological objectives. Rather than needing more research into the ecological effects of fire, the real challenge lies in implementing effective fire management. The key issues include:

- Reconciling conflicting goals, including the need to ensure human safety while simultaneously allowing fires to effectively rejuvenate vegetation, or finding ways to retain the benefits of plantation forestry without fueling fire-driven invasion by alien trees.
- Finding effective ways to deal with intractable problems, such as invasion by fire-adapted alien trees and shrubs in a fire-prone environment.

 

- Agreement on acceptable trade-offs in cases where all goals cannot be met, including increasing management efforts in priority areas (Forsyth *et al*. 2012) while reducing them elsewhere, or balancing the need to create labor-intensive work (necessary to gain political support and funding) with the need to invest in monitoring and evaluation that will employ fewer people, but will increase efficiency (van Wilgen *et al*. 2012b).
- Building management capacity so that there are adequate numbers of trained managers, and established protocols and procedures, to support professional fire management.
- Obtaining high-level and sustained support for adaptive management, which will require long-term monitoring and assessment programs that may take a considerable time to produce tangible results.

The final point above is especially important. The adoption of adaptive approaches to management and implementation of a culture of goal-setting, assessment, and continual improvement offer the best chances for success, and this will require sustained commitment at high levels.

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